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Monitoring and Data Analysis to Support Adaptive Site Management

Adaptive site management (ASM) is dependent on the the development of analytical tools to help site managers determine when, and to what degree, a change of remedy will better achieve the goals of cleanup. At the same time, these tools should help demonstrate to diverse stakeholder groups that changes are warranted. It is important to gain support from the affected public and from public or private transferees prior to making changes in remedial strategies, even when an agreement has already been reached between the lead regulatory agency and the responsible party. Consensus can best be achieved if there are objective methods that help evaluate the potential changes.

This chapter considers analytical tools and monitoring techniques that can aid in the assessment of remediation performance and help site managers decide if the current remedy-in-place should be reevaluated. Monitoring programs supply the information required to support the four management decision periods (MDP) described in Chapter 2. For example, analysis of monitoring data is needed to determine whether performance standards and operational expectations have been met, whether remedial goals have been achieved, and ultimately whether site closeout can occur.

ANALYTICAL TOOLS FOR EVALUATING REMEDY EFFECTIVENESS AND NEED FOR CHANGE

Both graphical and tabular techniques exist to help make decisions about the effectiveness of remedies and the need for change. Tabular methods attempt to characterize the various objectives and attributes of interest for alternative remediation plans and display them on a single table so that they may be considered together. These objectives could include human health and ecosystem risks (or risk reductions), contaminant mass remaining (or removed), projected time and cost to completion of remediation, projected land use and property values at or near the site, and a qualitative indication of the likely extent of support or opposition among different stakeholder groups. This presentation should help illuminate major advantages and disadvantages of each alternative, and indicate the tradeoffs between the desired objectives that occur in switching from one remediation plan to another.

More formal analysis is also possible using various techniques of multiattribute utility theory (Keeney and Raiffa, 1976; Keeney, 1980; Merkofer and Keeney, 1987; Edwards and Barron, 1994; Clemen, 1996; Farber and Griner, 2000). Examples include the assignment of weights to different objectives (both by the site manager and by different stakeholders) to see how sensitive preferred alternatives are to these differing weights. As a hypothetical example, the eight objectives identified in Chapter 2 could be used, with differential weight being given to them to reflect laws and regulations and stakeholder preferences. The outcomes of different remedies can be ranked in an attempt to identify the most promising alternative. Such techniques have been employed to help facilitate stakeholder deliberations and decisions for other environmental management problems (Jennings et al., 1994). Often, such deliberations are best supported with simple and effective graphical presentations for each alternative, as discussed below. One weakness of this approach is that it can be difficult and costly (in terms of time and resources) to obtain quantitative values for all objectives.

In addition to the tabular approaches, a number of graphical options can be developed to illustrate when changes in a remedy might be necessary. For remediation operations based upon contaminant extraction (e.g., pump-and-treat or soil vapor extraction), the most straightforward graph would be one that displays mass removal over time, as shown in Figure 3-1. Indeed, such graphs are already commonly prepared in practice, as discussed in Chapter 2 in the Lawrence Livermore case study (and other case studies described later). Recent Navy guidance (NAVFAC, 2001) advocates preparation of performance plots of monthly operation and cost data similar to Figure 3-1.

Although mass removal is one objective measure of the remediation performance, cleanup goals are normally based upon reduction of total pollutant concentrations to health-based standards. [Such cleanup goals contain an implicit assumption that total concentration levels determine risk, which may or may not be accurate depending on the bioavailability of the contaminant (NRC, 2003)]. Therefore, another way to assess the progress of remediation is to plot the temporal changes in concentration at chosen "sentinel" monitoring wells (e.g., wells located at the down-gradient property boundary or adjacent to critical receptors). Such a plot is represented by Figure 3-2, which shows both hypothetical contaminant concentration over time as well as the *reduction* in contaminant concentration (or reduction in risk) over time. This second measure is more reliable, because calculation of the baseline risk associated with the initial contaminant level is fraught with uncertainty, whereas there is less uncertainty about the risk *reduction* (as measured by the surrogate concentration reduction).

The hypothetical graphics shown in Figures 3-1 and 3-2 are drawn to represent a single remediation technique (e.g., pump-and-treat, soil vapor extraction). Analogous curves using different measured parameters could also be drawn to describe containment technologies (e.g., sediment capping), which aim to limit the contaminant mass flux through a "compliance" boundary. Of course, a containment technology would have

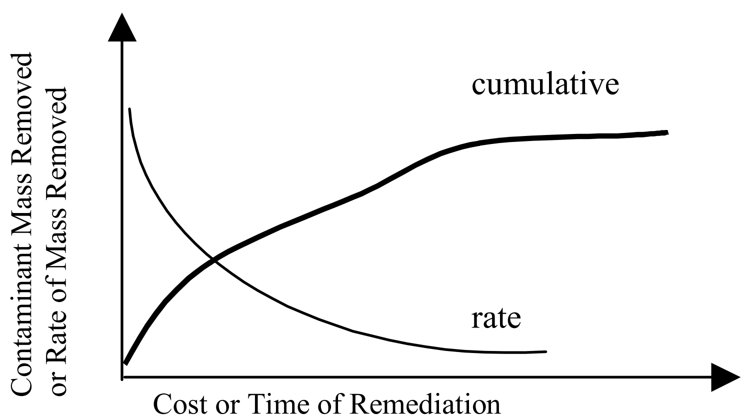

FIGURE 3-1 A hypothetical plot of contaminant mass removed over time or over cost, for a remedy based on extraction of mass. Both the cumulative mass removed and the rate of mass removed are shown.

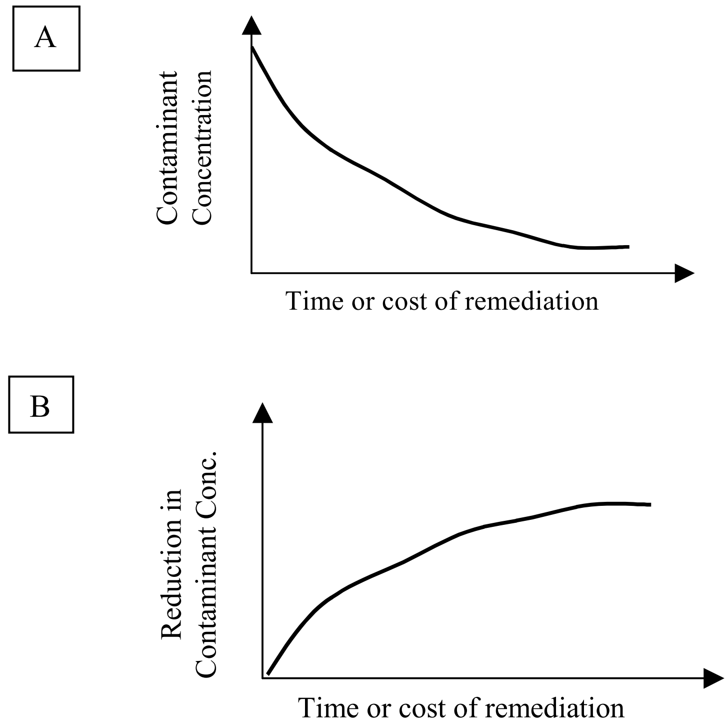

FIGURE 3-2 Hypothetical plot of (A) contaminant concentration over time or over cost and (B) reduction in contaminant concentration over time or cost.

little or no impact on mass removal (Figure 3-1), but would achieve dramatic reduction in risk; this is an example of an exposure reduction ("E") strategy as previously discussed in reference to Figures 2-1 and 2-2.

Ideally, there would be a set of performance curves like those in Figures 3-1 and 3-2 for different remediation methods or management options such that the curves could guide decisions as to which option to select and when to change from one approach to another. As an illustration of such curves, consider Figure 3-3, which shows a family of hypothetical curves for the risk reduction over time for various types of remediation systems. Curves A, B, C, D, E, and F within Figure 3-3 suggest a wide range of potential results from different remedies. Attaching specific strategies to any given curve is not possible without more information on the type of contamination, the predominant exposure pathway, and the affected receptors. However, one can speculate that Curve A represents a mass removal strategy such as *in situ* chemical oxidation of dense nonaqueous phase liquids, where a high percentage of mass must be destroyed before a significant reduction in groundwater concentrations and thus risk is achieved (see Box 5-12 for more explanation of this behavior). Curves B, C, and E could represent any number of strategies where risk is reduced incrementally over time from the source zone, including monitored natural attenuation. Curve F may represent a strategy like containment or a landfill cap where no contaminant mass is reduced, with the dotted line representing the possibility of future catastrophic failure.

The "effectiveness" of any particular remedy could be based on the ratio of risk reduced per unit of time or cost. (Keep in mind that it is difficult to quantify risk, and thus the ordinate axis may actually represent reduction in concentration.) Higher ratios would be desirable, and any remedy that provided the higher ratios may be considered well suited for the particular risk reduction goal. Lower ratios would suggest that either the remedy is not appropriate for meeting the risk reduction goal or the remedy needs to be optimized.

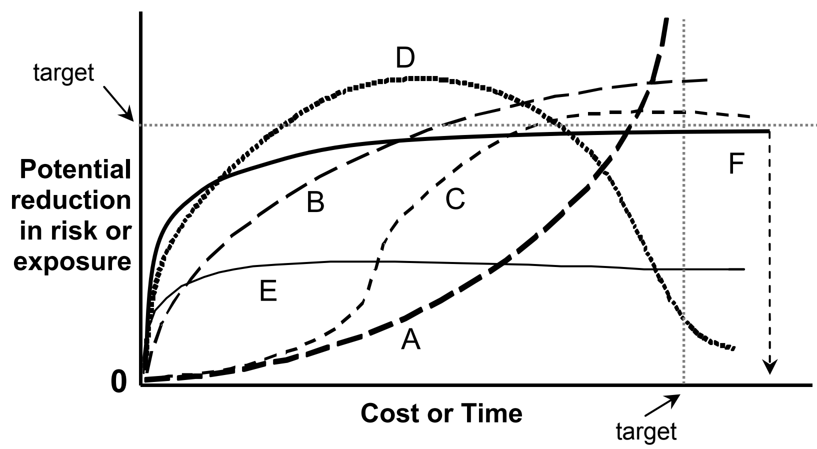

FIGURE 3-3 Hypothetical graphical representations of the change in risk with time or cost for different remedies.

Of these curves, only Curve E suggests a remedy that is totally ineffective in meeting cleanup goals and perhaps would provide the strongest graphical illustration of the need to change or modify the remedy. Clearly, remedies for which costs are increasing without any noticeable reduction in risk should not be continued. The case for change may be less clear for Curve A, which will eventually result in risk reduction albeit at longer time periods and higher costs than Curves B and C. The shape of Curve B is analogous to several case studies (see Box 2-3 and Appendix B) in that there is a relatively large initial reduction in contaminant concentration followed by a long period of relatively small reduction. Curve D represents a unique case in the sense that risk is seemingly being reduced quite effectively, yet as the remedy continues longer, the risk increases. This type of result may occur when source materials are drawn into an area or aquifer as a result of the remedy, increasing the concentrations of the contaminants to such a degree that higher risk results. This may also be the same type of curve that would result when an effective remedy is turned off and a rebound in concentrations occurs as the plume continues to move through the monitoring wells (but only if time, not cost, is the x-axis; if you turned off a remedy, presumably the cost disappears).

In addition to the qualitative assessments of the various curves described above, graphical tools could provide more quantitative guidance, assuming that reliable and accurate values for cost and risk reduction can be measured. For example, if there is a desired target goal for risk reduction, then a horizontal line can be drawn from this target to find the "least cost" remediation scheme. Using the example illustrated in Figure 3-3, Curve B would be conceptually the most desirable over the mid term, although Curve D achieves the target risk reduction at the least cost over the short term. Conversely, if there is a target remediation budget, then a vertical line can be drawn from this target to find the most effective remediation scheme. Using the example illustrated above, Curve A would be selected.

These examples are intended to be illustrative, and more detailed quantitative assessments are possible. For example, the slopes of the curves in Figure 3-3 measure the marginal risk reduction per unit investment, and these can be used in principle to optimally switch from one curve to another. Of course there may be other constraints that preclude such flexibility, and the difficulties in generating the risk reduction estimates must also be appreciated.

As discussed in Chapter 2, risk reduction may not be the sole objective of a site remediation strategy. For example, if both contaminant mass removal *and* risk reduction objectives are sought, then the problem becomes more complicated to visualize; however, graphical tools such as the one illustrated in Figure 2-2 could be developed. In this case each remediation system is represented by two curves, one measuring its performance for the risk reduction objective, and the other for the mass reduction objective. The time horizon for remediation is another objective that is often not considered explicitly during the remedy selection phase. However, short remediation times would be highly desirable in scenarios where the property is to be transferred for economic development. In most cases a single remediation strategy will not be capable of simultaneously satisfying all the objectives. The value of such a multidimensional graphical plot is that tradeoffs among objectives and strategies become evident, thus establishing a framework for stakeholder input and negotiation.

Although development of performance curves is advocated in recent Navy guidance (NAVFAC, 2001), they are not routinely developed at most sites, particularly for soil and groundwater contamination. Rather, the general sequence of events is to determine a remedial goal and then choose a technology that will meet the goal at lowest cost. For sediment contamination, it is more typical to use the type of predictive models that could generate these performance curves in choosing the remedy (e.g., see Figure 3-4). The committee strongly recommends that the Navy make a concerted effort to collect the appropriate performance data so that these curves can be generated for various types of remedial actions and hydrogeologic settings. Indeed, data likely exist from Department of Defense (DoD), Department of Energy (DOE), and Superfund sites, as well as from government demonstration programs like the Environmental Security Technology Certification Program. The goal is to develop a set of models for broad classes of remedies, contaminants, exposure pathways, and receptors that can then be calibrated (most logically during the feasibility study) with site-specific data to generate performance curves applicable to a specific site. Developing the models in the first place will require data collection at sites where remedies are already in place, including data on contaminant concentrations at compliance or receptor locations if risk reduction is a desired metric. The benefits of this exercise are accrued later when the resulting models are calibrated with site-specific information and then used to inform remedy selection. Because such models reflect our current understanding of subsurface processes, which in some cases is limited, the models should be updated as performance monitoring data become available.

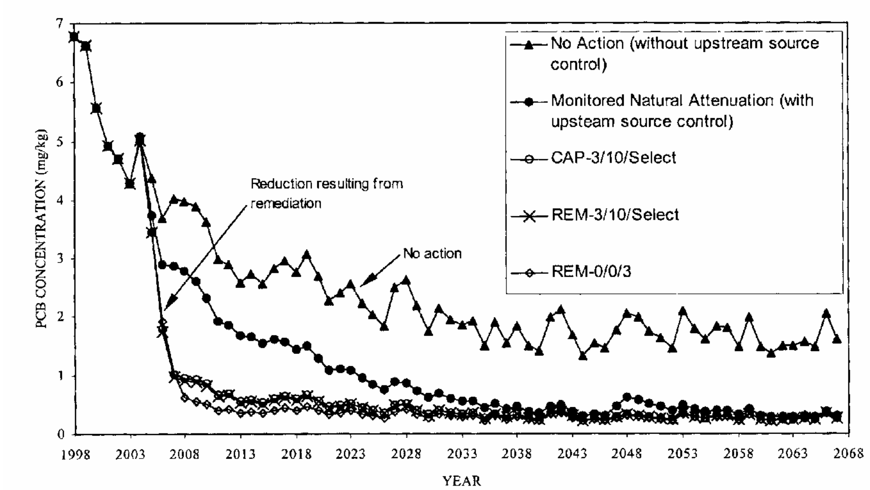

FIGURE 3-4 Model projections for polychlorinated biphenyl (PCB) concentrations in Thompson Island fish from 1998 to 2068 for various remedial alternatives as outlined by EPA Region 2. The noise in the "no action" projection is due to year-to-year variability in the projected flow record, which reflects the statistics of historical flows. SOURCE: Reprinted, with permission, from the National Research Council (2001). © (2001) National Academies Press.

Graphical tools can also be used to make decisions after implementation of a remedy, particularly in conjunction with the specific management decision periods of ASM. In addition to answering the three questions of MDP2 (is the remedy meeting performance standards, is it meeting operational expectations, and is it meeting the remedial goal), graphical analysis of monitoring data can enable identification of asymptotic conditions where concentrations are not low enough at the site to achieve the health-based remedial goal, and operation and maintenance costs have become high enough to raise concerns. Interpretation of the graphs to provide yes or no answers to these questions will be subjective, because there will likely be disagreement about various critical performance criteria (e.g., at what dollar value does the cost per pound removed become cost-inefficient, or at what slope of the concentration versus time curve should the remedy be changed). Nevertheless, the graphs will indicate trends that provide information needed by the remedial project managers (RPMs), regulators, and stakeholders for decision making.

Several case studies already exist demonstrating how graphical tools can aid in making decisions to modify remedies and in evaluating remedial objectives. In almost all these examples, concentration is the measured parameter and is used as a surrogate indicator of risk. The first study is from the set of volumes published by the U. S. Environmental Protection Agency (EPA) under the auspices of the Federal Remediation Technologies Roundtable (EPA, 1998a). At Pope Air Force Base, as much as 75,000 gallons of JP-4 free product are floating on top of the water table; some dissolved volatile organic compounds (VOCs) have also been detected in groundwater samples. The remediation system consists of a free product cut-off trench and a dual pump recovery system. Figure 3-5 shows a decreasing removal rate over time for free product at this site, such that the cumulative recovery curve begins to flatten after April 1995. (Note that the EPA report includes another case study for a different free product removal site at Pope AFB where the cumulative removal continues to increase approximately linearly over time.) As of the last reported date (October 1996), approximately 3,500 gallons had been recovered, less than 5 percent of the estimated spill volume of 75,000 gallons.

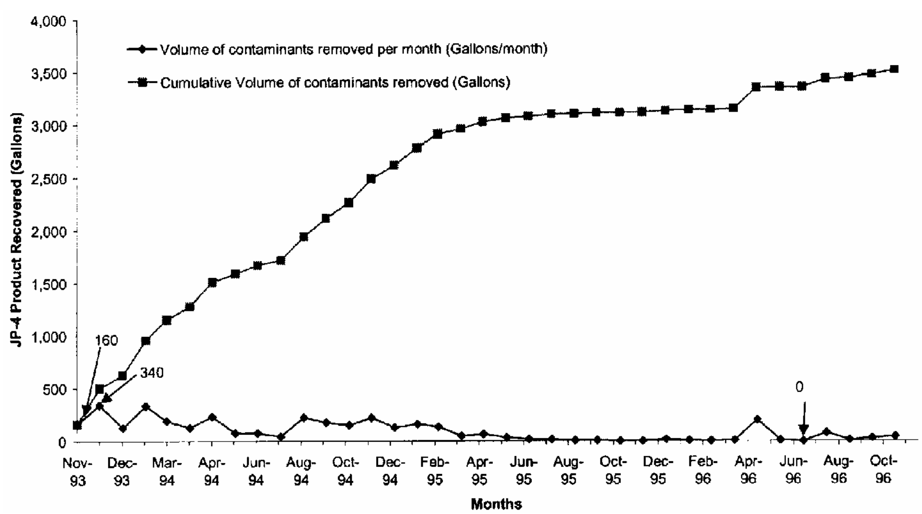

FIGURE 3-5 Monthly and cumulative free product removal at Site SS-07, Pope AFB. SOURCE: EPA (2000a).

An interesting aspect of this case study is that estimates of operation and maintenance (O&M) costs are combined with the above data to produce a graph showing cumulative costs versus cumulative pollutant mass removed. This graph, presented in Figure 3-6, illustrates the economic impact of the "tailing" behavior—as the remediation progresses, it becomes increasingly more costly to remove a given unit of contamination. However, it should be noted that Figure 3-6 was produced by making the simple assumption of constant average monthly O&M costs. Because the monthly costs are constant, and the monthly removal rate decreases over time as shown above, the cost per unit gallon removed will increase. This graph indicates that the performance of the remediation system has declined because little additional mass is being removed as funds continue to be spent, signaling that the system should be reevaluated. A reevaluation may or may not lead to a change in remedy, depending on the expected performance of the technology and other factors.

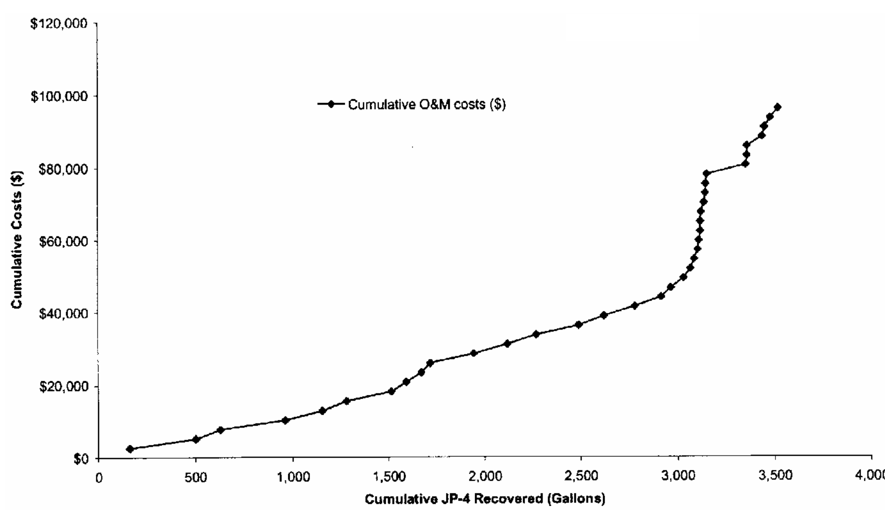

FIGURE 3-6 Free product removal versus cumulative operation and maintenance costs at Pope AFB. SOURCE: EPA (2000a).

A second case study is for the Campbell Street Fuel Farm groundwater pump-and-treat system located at Marine Corps Air Station New River, which is co-located with Marine Corps Base Camp Lejeune, North Carolina (NAVFAC, 2001). The fuel farm is an active storage facility for JP-5, and release of fuels at the site has led to contamination of soil and groundwater by benzene, toluene, ethylbenzene, and xylene (BTEX) and VOCs. Contaminated groundwater is limited to the upper portion of a surficial aquifer with its water table 6 to 7 feet below ground surface. Initial remedial actions at the site were excavation of contaminated soil and removal of measurable free product. A groundwater pump-and-treat system began operation in July 1996; the system includes interceptor trenches and several extraction wells that were installed in plume hot spots. The trenches are downgradient of the contaminant plume, and all intercepted water is directed toward sumps for removal.

Figure 3-7 shows that the VOC mass removal rate has decreased significantly over time; while 3.5 pounds were removed during July 1996 through March 1999, less than 0.5 pounds have been removed since December 1997. Figure 3-8, a plot of the cumulative cost versus cumulative mass removed, dramatically displays the tailing behavior of the system. It can be seen that approximately $175,000 was spent to remove the first 3 pounds of VOCs, but an additional $325,000 was spent to remove the next 0.5 pounds. The graphical data below were used in conjunction with other analyses and assessments to recommend that the trenches be shut down and that monitoring data be collected to evaluate the degree to which the plume was being affected by natural attenuation processes. Figure 3-8 suggests that caution and knowledge of the chosen treatment are needed when interpreting such graphs for the purpose of making changes to the remedial system (as discussed in Chapter 2 with respect to MDP2). It would have been premature to abandon the pump-and-treat system at the first sign of cost inefficiency in late 1996. Fortunately, site managers recognized that such systems generally take years before performance reaches an asymptote; continued operation resulted in a substantially longer period of effective mass removal.

When the graphical tools indicate the remediation system should be reevaluated, changing the remedy can improve the system, as illustrated graphically in Figure 3-9. This figure schematically depicts contaminant concentration versus time when changing from a suboptimal remedy (such as that depicted by Curve E in Figure 3-3) to another remedy (Curve B in Figure 3-3). Changing the remedy should alter the concentration versus time curve such that the target contaminant level is reached sooner.

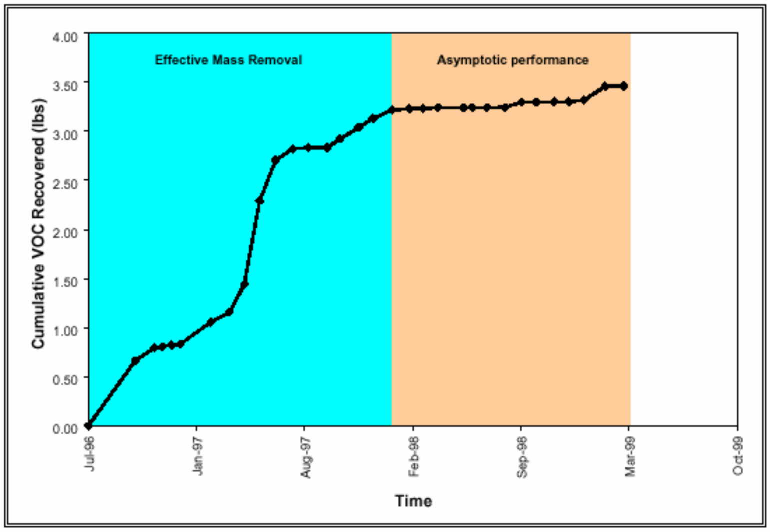

FIGURE 3-7 Cumulative mass recovered versus time for the pump-and-treat system at the Campbell Street Fuel Farm. SOURCE: NAVFAC (2001).

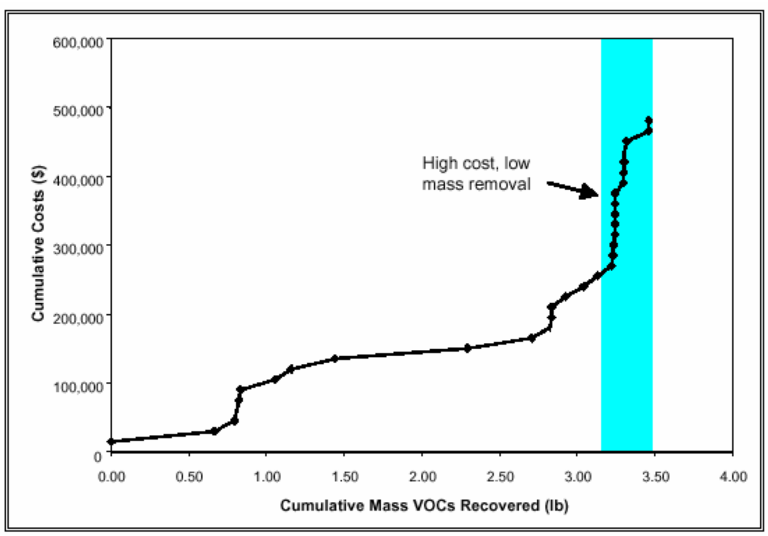

FIGURE 3-8 Cumulative costs versus cumulative mass of VOCs removed for the pump-and-treat system at the Campbell Street Fuel Farm. SOURCE: NAVFAC (2001).

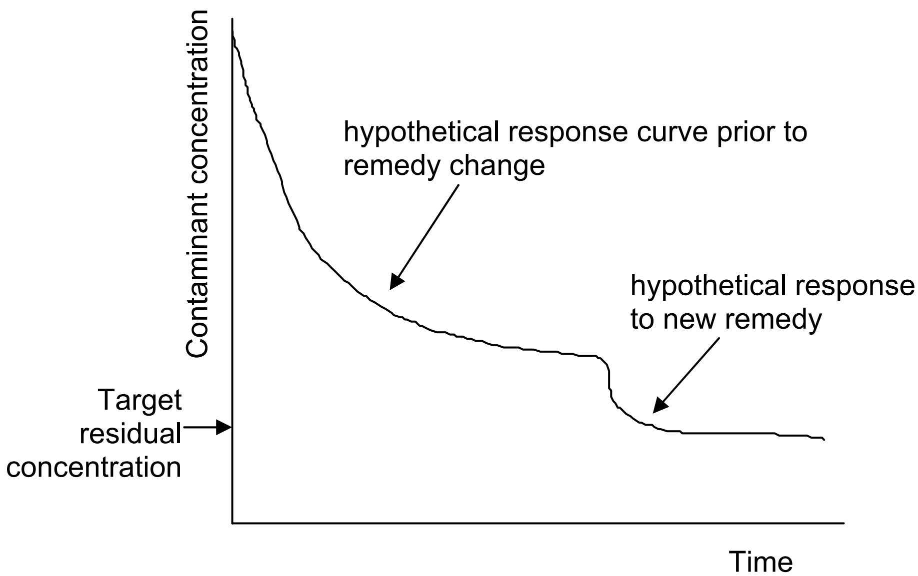

FIGURE 3-9 Hypothetical effect of changing the remedy on the concentration versus time curve.

Despite the problems that may arise if data are highly variable, there is merit in undertaking the graphical approach described above provided that the data reveal trends. In general, the remedy should be revisited when there are reasonable quantitative data showing that the existing remedial action cannot attain the health-based remedial goal selected for the site after being operated for an appropriately long period of time. In order for this to occur, the slope of the line tangent to the concentration versus time curve must be approaching zero (the so-called asymptote), yet the concentration must remain above the site-specific remedial action goal. Second, it should be shown that the cost of treatment is increasing sharply as the incremental mass removed decreases, even though the annual costs may remain constant. Whenever possible, visual interpretations of these data should be supported by statistical analyses to ensure that the inferred value of a trend (or the lack of a trend) is statistically significant. [Overviews of statistical methods for trend analysis are found in Lettenmaier (1976, 1977), Gilbert (1987) and Gibbons and

Coleman (2001)]. Simple methods include cumulative sum charts that accentuate the long-term effects of trends and changes in the mean (Berthouex et al., 1978), and linear regressions of concentration versus time from which it can be determined whether the slope of the fitted regression model is significantly different from zero. Methods for data quality control can also be adapted for trend analysis (Starks and Flatman, 1991). With large datasets, methods for time-series analysis can be used to identify and remove trends and statistical periodicities in the data over different time scales (Box and Jenkins, 1994). Nonparametric methods may also be employed for trend detection; these are especially appropriate when, as often occurs with environmental data, the variations in the measurements are not normally distributed.

When cost and concentration data analyses reveal declines in remedy performance prior to reaching the cleanup goal, the responsible party should undertake reconsideration of the remedy with the same public participation steps that are utilized in the original remedy selection process. In order for these exercises to be effective, the Navy, in consultation with stakeholders, should select a unit cost for the continued operation of the remedial action at the site under investigation, above which the existing remedy is no longer considered a tenable option. This value will necessarily vary from site to site to reflect the type of technologies used, site conditions including the existing contaminant concentrations compared to the cleanup goal, the toxicity of the contaminants, the likelihood of future exposure, and other factors. It is possible that there will be some regulatory and stakeholder reluctance to using a metric such as “cost per pound of contaminant removed” for decision making. Members of the public are often suspicious of risk assessment in general (see Box 2-2), particularly attempts to place a monetary value on individual lives and public health. Nevertheless, in the committee’s experience, most community activists react constructively when given pertinent technical and financial information and the chance to fully participate in decision making. Typically, if incrementally more cleanup can be demonstrated to make the local environment significantly safer, most stakeholders will insist upon the higher-level response. If additional actions will only marginally improve safety, and this can be conveyed using the types of graphical presentations discussed previously, stakeholders will give it due consideration (as was evidenced by community activist sentiment regarding a mercury-contaminated site in Oak Ridge, TN—NRC, 2003). Graphs showing predicted and real performance curves (and other evidence that responsible parties and regulators have the public’s interests in mind) are also more likely to make the public receptive to limited cleanup at certain locations if other sites receive *greater* attention as a result of the same type of analysis.

Consideration of Uncertainty

The discussion of tabular and graphical tools above neglects the inherent uncertainties that are present in risk assessment and performance assessment of groundwater remediation. Uncertainty is a significant reality associated with all environmental monitoring programs and is the result of (among other things) limited spatial and temporal data from which inferences must be drawn. Uncertainty is particularly prevalent in our understanding of subsurface properties, including stratigraphy, presence or absence of preferential flow paths or fractures, porosity, hydraulic conductivity, and boundary conditions. There is also substantial uncertainty at a given site regarding the nature and extent of contamination, the type of biological and geochemical processes that might be taking place that affect contaminant fate and transport, and the exposure mechanisms that translate into deleterious effects (NRC, 1999). As a result, there is significant uncertainty associated with any estimated contour map of a contaminant plume as well as the total contaminant mass. There may also be significant uncertainty about whether the measured total mass of a contaminant in the subsurface is directly correlated with exposure or risk. Because of these uncertainties, it is not possible to assign a single value to either the baseline risk, or to the risk reduction that could be achieved by a given remediation technology.

The extent of uncertainty about site conditions and remedial performance has implications for decision making throughout ASM. For example, at MDP2 the uncertainty in performance monitoring data plays a significant role in determining whether cleanup goals are being met. Mass removal achieved by ongoing remediation (Figure 3-1) is generally known to a high degree of certainty, but the critical factor is how close the asymptotic cumulative value is to the total pollutant mass at the site. In many cases, it may not be known whether the curve is leveling off at 5 percent, 50 percent, or 95 percent of the total (but unknown amount of) onsite contamination. The uncertainty may be particularly high at complex sites with a high degree of heterogeneity, multiple aquifer layers, fractured rock, and/or the presence of nonaqueous phase liquids (NAPLs) that can move in unusual ways from source zones, or remain entrapped at disparate locations on- or off-site. Similarly, river or coastal sediment beds with unusual hydrologic and sediment transport and deposition/resuspension regimes can lead to a high degree of uncertainty in the quantity and location of remaining contamination following cleanup efforts.

The overall uncertainty in the total mass onsite is shown schematically in Figure 3-10. There, M_t denotes the unknown total contaminant mass in the system, and the double-headed arrow is meant to convey uncertainty in that value. (Note that there could also be uncertainty in the asymptotic value of the cumulative mass removed.)

Uncertainty can also be represented on graphs that plot the reduction in contaminant concentration as a function of time or cost of remediation (such as Figure 3-2). These data could be generated from monitoring at chosen compliance or sentinel wells. However, because of the inherent spatial variability of contaminant fate-and-transport processes, there will always be uncertainty about the contaminant levels in portions of the site that are not monitored. Moreover, there are uncertainties that arise in computing human or ecological risk from ambient groundwater concentration values, given a lack of knowledge about how much of the total contaminant concentration is actually bioavailable. Incorporating this uncertainty into the graphical representations of concentration and risk reduction in Figure 3-2 and 3-3 is even more challenging.

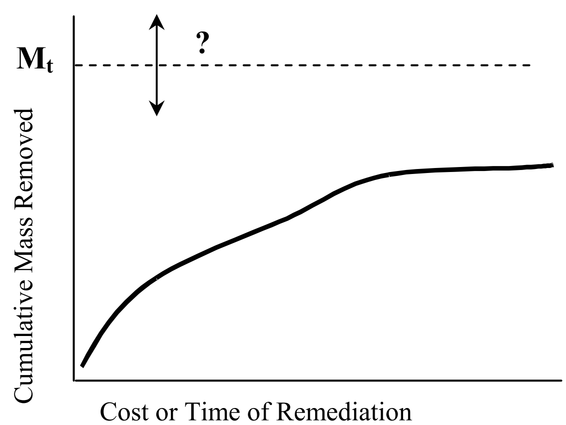

FIGURE 3-10 Hypothetical graph of cumulative mass removed over cost or time, showing uncertainty in the value of the total mass present (M_t).

One possible graphical technique is the concept of statistical confidence bands. The concept assumes that the risk (or concentration) reduction achievable for any given cost is a random variable with a certain probability density function. The random nature results from all the uncertainties in the system—for example, uncertain initial contaminant mass, uncertain remaining contaminant mass for a given remediation cost, uncertainty in groundwater fate-and-transport models, and uncertainty in dose–response models. The use of confidence bands is demonstrated conceptually in Figure 3-11. For any given remediation cost, the confidence bands could represent, for example, the 5 percent and 95 percent probability levels. That is, there is a 95 percent probability that the risk reduction is less than the upper curve, a 5 percent probability that the risk reduction is less than the lower curve, and thus a 90 percent probability that the risk reduction is between the upper and lower curves. The solid center curve might represent the mean or "best estimate."

The figures discussed above are for treatment-based remediation strategies where there is a direct correlation between performance and time. Strategies based upon exposure pathway intervention, such as sediment capping, onsite containment, or institutional controls (see the "E"-type strategies discussed in relation to Figures 2-1 and 2-2), perform in either a success or failure mode. Thus, performance uncertainty involves mainly the time to potential failure and, to a lesser extent, the nature of the failure (e.g., catastrophic or gradual). This is illustrated schematically in Figure 3-12.

There is an increasing body of literature that presents ideas along the lines discussed here, especially the concept of formally incorporating uncertainty into remediation design (mostly pump-and-treat). The main types of uncertainty considered are related to site hydrogeology. A typical statement of the design problem is as follows: design a remediation system that is guaranteed to work with a probability of at least X percent. Tradeoffs between reliability and cost are developed by varying the success probability level. A recent review of this work is given by Freeze and Gorelick (1999). Some more recent work (Minsker and Smalley, 1999) is extending these design concepts to be based more directly upon human health risk. Although most published work emphasizes development of the methodology with application only to hypothetical scenarios, Russell and Rabideau (2000) present an application to an actual site near Buffalo, New York.

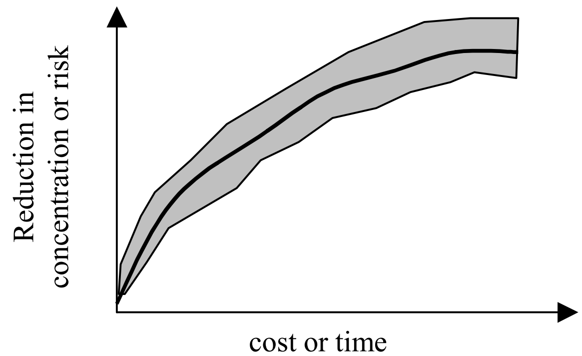

FIGURE 3-11 Statistical confidence limits around the curve of concentration reduction over cost or time. The upper and lower curves correspond to the 5 and 95 percent probability levels.

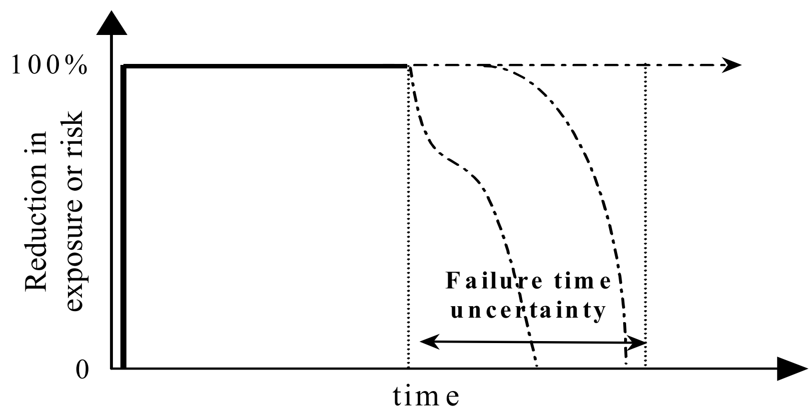

FIGURE 3-12 Statistical confidence limits around the time of potential remedy failure.

The work of Maxwell et al. (1998) (initially discussed in NRC, 1999) further illustrates formal uncertainty analysis concepts. The significant feature of Maxwell's work is that it combines uncertainty about groundwater fate and transport with variability in human receptors due to factors such as body weight and daily habits of water consumption and vapor inhalation. Typical results show the probability of increased cancer risk for different fractiles of variability in the receptor population given an exposure pathway of drinking contaminated groundwater. In more recent work, Maxwell et al. (2000) extend these concepts to evaluate the impact of different pump-and-treat remediation systems on reducing risk for a hypothetical contamination scenario. Their results do show that remediation reduces risk but, interestingly, there are differing amounts of risk reduction to different segments of the population. Their results are presented in the form of Figure 3-11, with confidence bands added to reflect fate-and-transport uncertainty. In order to consider variability among receptors, different curves (each with different confidence bands to reflect uncertainty) are drawn for different members of the population. Figure 3-13 provides an example, in which the curve corresponds to one segment of the receptor population, and the vertical bars indicate the uncertainty (approximate confidence limits) that is due to geological variability for the two different pumping rates that were studied.

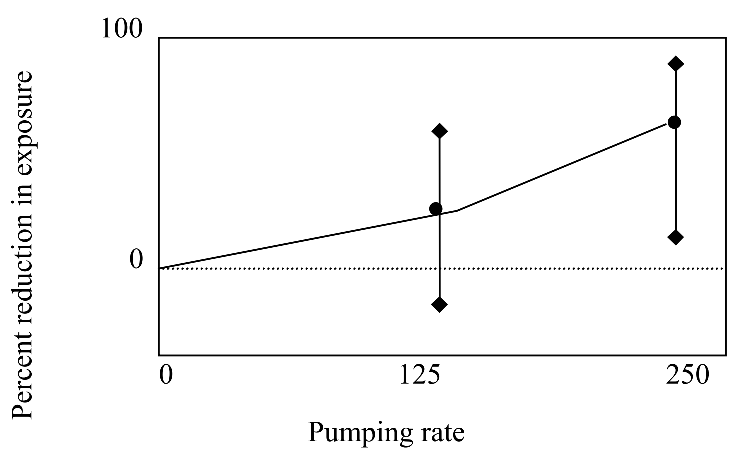

FIGURE 3-13 Uncertainty in the percent reduction in exposure achieved as remediation efforts are changed by varying the pumping rate. SOURCE: Maxwell et al. (2000).

In another interesting recent contribution related to contaminated sediments, Stansbury et al. (1999) present a methodology for accounting for uncertainty in human health risks, ecological impacts, and remediation costs of different strategies for disposal of dredged material. The method is demonstrated using an example of contaminated sediment disposal at Elliott Bay, near Seattle, where the possible remedial alternatives are (1) unconfined aquatic disposal (UAD), (2) capped aquatic disposal (CAD), (3) near-shore confined disposal facility (CDF), (4) upland disposal (UPL), and (5) upland secure disposal (UPS). Unconfined aquatic disposal is open water discharge of the dredged material. Capped, or confined, aquatic disposal is open water discharge of the dredged material into a prepared or existing depression in the sediment and capping with clean sediment. A near-shore confined disposal facility is an in-water landfill, generally with only primary sedimentation as treatment during placement. Upland disposal and upland secure disposal are both conventional landfills, the first with simply primary treatment and the second with more elaborate containment. These remediation alternatives achieve their effectiveness at the time they are implemented, whereas most groundwater remediation alternatives need to operate over extended time periods.

Stansbury et al. (1999) do not use probabilistic techniques but rather adopt the formalism of "fuzzy set" methods to incorporate uncertainty. In this approach, uncertainty is represented by a range of "plausible" and "most likely" parameter values; this range can be established using a variety of information sources including measured data and engineering judgment. An example result is shown below in Figure 3-14, which shows tradeoffs among human health risk, uncertainty, and cost for the five remedial alternatives described above. For each alternative, the inner rectangle represents "high confidence" while the outer rectangle is still plausible but with lower confidence. The results show that upland secure disposal provides the greatest human health benefit, but at a very high cost. There is also a relatively large degree of uncertainty in the human health risk estimate; i.e., a given disposal strategy may provide relatively low cancer risk under one set of assumptions, yet it may also result in a high risk under a different, yet plausible, set of assumptions.

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NRC (1999) identified various ways to approach uncertainty in hazardous waste cleanup. In the face of limited information that typifies many sites, the use of conservative cleanup goals has been prevalent.

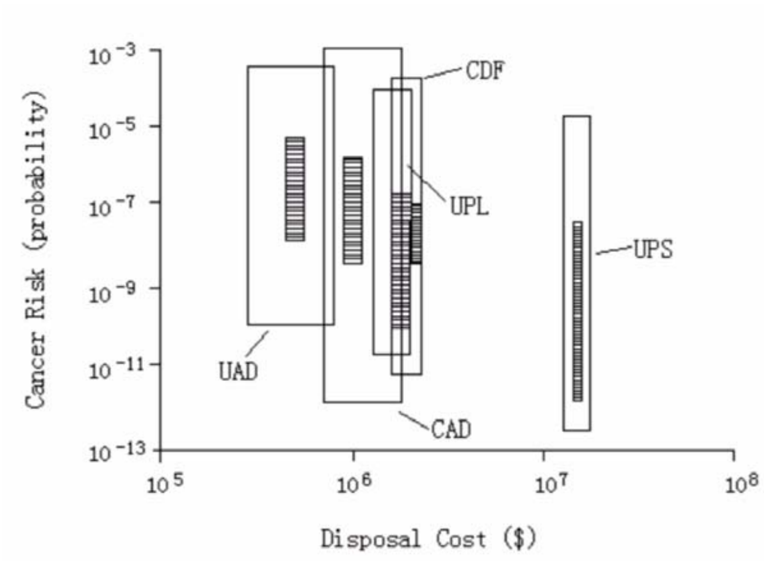


FIGURE 3-14 Tradeoffs between lifetime cancer risk and disposal costs for five disposal alternatives. For each alternative, the outer box denotes the range of plausible values, while the inner hatched box denotes the range of most likely values. UAD = unconfined aquatic disposal, CAD = capped aquatic disposal, CDF = near-shore confined disposal facility, UPL = upland disposal, and UPS = upland secure disposal. SOURCE: Reprinted, with permission, from Stansbury et al. (1999). © (1999) Journal of Water Resources Planning and Management.

Alternatively, attempts have been made to develop more comprehensive programs of site monitoring and characterization by, e.g., increasing the number of monitoring wells. Clearly there are distinct tradeoffs between these two approaches, both in terms of the information gathered and cost. For example, the use of only one monitoring well located within a contaminated area would require significant extrapolation as to what is occurring at more distant edges of the plume. This may result in a conservative approach to operating a pump-and-treat system even if the well yields consistent results below cleanup criteria. In contrast, a monitoring system that plasters an affected plume with sampling points would presumably be much more precise in its determination of when a system is in compliance and hence when remediation can stop, although it will also be more costly.

Because many remediation systems are overdesigned to account for uncertainties (i.e., via an engineering "safety factor"), there may be significant economic value in collecting data to assess and reduce uncertainties in remedial performance. It was for this reason that NRC (1999) favored the more rigorous data collection approach over the use of conservative goals. The formal uncertainty analyses presented above provide substantial benefits to such data collection efforts. First, such analyses are valuable aids for site decision making because they provide a graphical display of the variability and uncertainty that are inherent features of any remediation problem, which can be used, among other things, for communicating information to stakeholders (including the extent of confidence in predicted and actual remedial performance). Moreover, by analyzing and ranking the various factors that contribute the greatest to overall uncertainty, it is possible to direct data collection activities that might reduce uncertainty toward the most critical parameters. In Stansbury et al. (1999), it was found that the rather large range in cancer risk shown in Figure 3-14 for all disposal alternatives was due mainly to the uncertainty in the dose–response relationship for the contaminants rather than to uncertainty in exposure pathways. This suggests that reducing uncertainty could be better accomplished by investing in additional research on dose–response relationships rather than by exploring other remedial options. Similarly, Russell and Rabideau (2000) used sensitivity analysis to examine the impact of various modeling assumptions upon management decisions. Several authors have studied how information obtained from specific data collection programs can be used to most effectively reduce uncertainty in contaminant fate-and-transport predictions and also in resulting site management decisions (e.g., James and Gorelick, 1994; Wagner, 1999; Sohn et al., 2000). It should be noted that such uncertainty analyses may only be feasible for larger, more complex sites where a fate-and-transport model is already available.

MONITORING

Monitoring plays a pivotal role at all stages in adaptive site management—from initial site discovery to site closeout. A cursory examination of Figure 2-7 might suggest that monitoring is needed only to answer the three questions posed during MDP2. However, monitoring programs are essential to facilitate site characterization and risk assessment (Step 1), to adequately conduct experimentation and evaluation, to produce the data necessary for constructing the performance evaluation graphs described earlier (which would be used during remedy selection or MDP3), and to determine whether residual contamination exists that will prevent site closeout during MDP4. The focus of the monitoring programs is necessarily site- and time-specific. For example, a soil remedial action may primarily require sampling during excavation (performance monitoring) and immediately after remediation work is complete (site closeout). For sediment and groundwater remedial actions, much longer-term monitoring programs might be developed that have their roots in initial site characterization activities, continue through remediation, and extend for significant periods of time beyond the termination of active remediation. In the case of groundwater, most sites begin with an inherited set of monitoring points already established, and so part of the monitoring design process also includes determining to what extent this existing network can be used or must be abandoned or expanded. Depending on the chosen remedial actions, monitoring programs may represent the majority of remedial action costs (such as for monitored natural attenuation) or only a small percentage.

The design of a remedial action performance monitoring network requires determining the parameters of interest, identifying the numbers and locations of monitoring points, specifying sampling protocols, frequencies, and analytics, and, finally, developing the data analysis methods that will support the decisions that have to be made. Traditional characterization and monitoring programs tend to pre-specify sample numbers, locations, sampling frequency, and analytics, where the emphasis for analytics has been on offsite laboratory analyses. This traditional type of data collection presents several limitations, particularly in the context of subsurface characterization and monitoring. The costs are sometimes prohibitive, driven both by sample analytical costs and the capital investment required for monitoring wells. High monitoring costs, particularly for monitoring programs that extend over time, result in pressures to limit data collection. Limited data collection, in turn, results in decision making that relies on datasets too sparse to adequately address the inherent heterogeneities and uncertainties associated with subsurface systems. Finally, by pre-specifying sample numbers and locations and relying on offsite laboratory analyses with long turnaround times for analytical results, traditional characterization and monitoring programs are ill equipped to handle unexpected results when they are encountered. Fortunately, in the last several years there have been technological advances in sensors, field analytics, and sample collection technologies that can help to lower costs and/or increase the effectiveness of monitoring programs. New approaches for designing and implementing environmental data collection programs have also been developed.

The following section discusses several different aspects of monitoring, starting with the parameters that are commonly used to measure remedy performance (relevant during MDP2, evaluation and experimentation, and long-term stewardship). The focus then shifts toward innovative monitoring network design that will facilitate use of ASM by allowing the entire remedial implementation period to be more adaptive. This includes discussion of new sampling technologies as well as ways to enhance existing sampling networks. The former is applicable to all stages of cleanup, from site characterization to long-term monitoring, while the enhancement of existing networks pertains primarily to long-term monitoring of contaminated groundwater. Thus, the case studies presented span various stages of cleanup, from initial characterization of a contaminated sediment site to optimization of groundwater monitoring. Indeed, because "site characterization" and "long-term monitoring" describe the same general activity—data collection with the purpose of understanding surface/sediment/subsurface contamination events at particular points in time—it should not be surprising that the same sampling technologies are appropriate for both characterization and later monitoring activities.

Monitored Performance Parameters

The performance evaluation graphs presented earlier focus on several key parameters measured over time. These include risk and risk reduction, contaminant concentration, contaminant mass removal, and cost. Aside from these primary parameters, there can also be a host of secondary, tertiary, and technology-specific parameters that might be included in a monitoring program. The section below discusses many of the most common performance parameters used for assessing remedy performance in contaminated soil, sediment and groundwater scenarios. To be rigorous, the monitoring system should provide multiple lines of evidence (as manifested by a variety of measured parameters) that a remedy is or is not effective.

Risk Parameters

Most cleanup goals in RODs are expressed as contaminant concentrations that correspond with a risk falling in the range of 10^{-6} to 10^{-4} for carcinogenic compounds. There are performance parameters that directly address risk without measuring mass or concentration reduction, such as growth and/or mortality of a target organism. Such parameters are measured in effects-based toxicity tests, and they are used primarily where cleanup is driven by ecological concerns because of the acceptability of performing these types of tests on plants and animals. For example, a suite of methods is available to assess toxicity of contaminants in soils and in freshwater and marine sediments to invertebrates and other animals, and newer methods that harness molecular biological techniques are being developed for high throughput toxicity testing of sediments (EPA, 2000b; NRC, 2002). Toxicity test results from a study area can be compared to those of samples taken in a reference area where the contaminants are absent or are present at reduced levels to determine whether toxicity in the study area is elevated above a level considered acceptable or shown to cause negative effects.

The use of such effects-based parameters raises two implementation issues that must be addressed. First, because of the time required for substantive results from remedial actions to be reflected by such measures, short-term measurements such as contaminant volume, mass, or concentration reductions will almost certainly be needed to supplement the long-term monitoring of toxicity. Second, there can be substances in the sediment or soil that cause a toxic response other than the contaminants of concern, making interpretation of results difficult. As a result, it is important to be familiar with the conduct of these tests, with the types of spurious results that might result in some sample types or matrices, and with how to interpret the data appropriately so that inaccurate conclusions are not made.

Indicators of Exposure and Risk

One of the main elements of risk is exposure, for which a variety of monitored parameters are indicative. Contaminant concentrations at key locations or in key media (e.g., in sentinel monitoring wells for groundwater, or in the overlying water column in the case of sediments) are commonly used and often codified in RODS, as mentioned above. It is important to differentiate concentration measurements that are direct indicators of exposure, such as water column, plant, invertebrate, or fish tissue concentrations, from total concentrations in soil and sediment, which, depending on the receptor and exposure pathway, may be more indirect indicators of exposure.

Remedial action performance monitoring programs almost always include *in situ* concentration monitoring as a significant component. Examples of this kind of monitoring include monitoring wells for groundwater and sediment sampling for contaminated sediments. The results are used to compare to concentration-based remedial goals, to develop spatially averaged concentration values, and to construct concentration isopleths. *In situ* spatially averaged concentration values, when combined with mass removal measurements (discussed later), allow both for a comparison with compliance requirements and for estimation of when these compliance requirements might be achieved. Concentration isopleths can be used to identify areas that are or are not in compliance with cleanup requirements.

In contaminated sediments not subject to physical disturbance like erosion, bioturbation—the mixing associated with the normal life-cycle activities of benthic organisms—is typically the most important mechanism for transporting contaminants to the sediment–water interface (Reible et al., 1991). Because more than 90 percent of the 240 observations of bioturbation mixing depths in both fresh and salt water were 15 cm or less and more than 80 percent were 10 cm or less (Thoms et al., 1995), surficial sediments are thought to be most important in contributing to exposure of (1) organisms in the sediment or overlying water and (2) animals that may feed off of these organisms. Isolated deeper penetrations by individual organisms apparently have limited impact on a population-wide basis. If only this surface layer contributes to exposure, then the surface area weighted average concentration (SWAC) in sediments presents a convenient monitoring metric. This metric has been employed as a measure of exposure and risk at several contaminated sediment sites—for example, within the ROD for the Sheboygan Superfund site and for the remedial investigation and feasibility study (RI/FS) at the Fox River site (Wisconsin DNR, 2001). It should be emphasized, however, that the biologically active layer is not necessarily static, and erosion can expose deeper sediments or deposition can bury surficial sediments with time.

Using sediment contamination as an example, a variety of direct and indirect concentration metrics can be used during MDP2. Thus, MDP2a (compliance monitoring) might seek to ensure that water quality standards are not violated during implementation of a remedial approach. MDP2b (monitoring to ensure that operational expectations have been met) could employ surficial sediment concentrations such as SWACs. MDP2c (monitoring to ensure achievement of remedial goals) might involve fish tissue concentration measurements.

In groundwater extraction systems, changes in contaminant concentration in produced fluid over time are a typical metric. The primary issue with this metric is that although extracted fluid contaminant concentrations are easy to measure, they are difficult to interpret from a performance perspective. For example, steady values of measured concentrations may mean that the system is performing well (particularly if these measurements can be linked to large mass extractions as planned). However, the same values may indicate a poorly performing system if levels are higher than cleanup goals. As with almost all of the metrics discussed in this section, contaminant concentrations need to be interpreted in conjunction with other remedial performance measurements.

Mass Removal

Although closure requirements are traditionally posed as either concentration or risk-based standards, in some cases cleanup is stated in terms of mass removal. Even in cases where mass removal does not necessarily translate into cleanup requirement compliance, it is obviously linked to attaining such standards. Thus, for remedial systems that physically extract and then remove or destroy contaminants, mass removal can function as a directly measurable performance parameter. Although this metric is less related to risk than concentration, mass removal is easy to measure and is not subject to spatial variability to the same extent as concentration. Mass removal is commonly measured for pump-and-treat systems and vapor extraction systems for groundwater and vadose zone contamination, respectively, and for excavation/dredging and disposal for soil and sediment contamination. Mass removal measurements are much more difficult for systems that rely on *in situ* processes to degrade or destroy contamination, such as *in situ* bioremediation or natural attenuation. The issues are twofold. Changes in concentrations at fixed monitoring points over time can be indicative of either degradation or simple transport and contaminant redistribution. Estimates of total mass degradation rely on interpolating from relatively sparse monitoring datasets to the system as a whole.

Specific metrics related to mass removal include the rate of contaminant mass removal. This rate could be measured in an instantaneous sense (i.e., the current rate of removal), or it could be measured in an aggregate sense (i.e., the rate of removal over the last quarter or over the last year). The latter, in particular, may be important for identifying a decline in performance over time. For systems where contaminant mass is physically removed and can be measured, implementing this type of metric is straightforward. For *in situ* systems, the challenge is obtaining accurate estimates of contaminant mass removal or destruction.

The percentage of total mass removed may also serve as a performance metric. The problem in implementing this type of metric is having an accurate estimate of the original contaminant mass; such information is frequently unknown. Sampling programs are discrete events in time and space, requiring inferences regarding spatial and temporal trends, often based on very limited datasets. For example, estimates of total *in situ* contaminant mass based on relatively large RI/FS datasets can be grossly in error, largely because the data gathering performed for an RI/FS is not intended and should not be assumed to be adequate to design the remedy. A site near Tonawanda, New York, had an estimated 14,000 cubic yards of contaminated soils. This estimate was based on 341 soil samples collected from 116 soil cores over a five-acre site during the RI/FS. By the time remediation was complete, 45,000 cubic yards of contaminated soils had been removed (Durham et al., 1999). Thus, it should be recognized by regulators, the Navy, and the public alike that additional sampling data will almost always be required after the RI/FS.

Secondary, Tertiary, and Technology-Specific Performance Parameters

Besides mass removal and *in situ* concentration, there can be a host of secondary, tertiary, and technology-specific performance parameters that might be included in a monitoring program. Examples of secondary and tertiary parameters include daughter products from bioremediation processes, pH, dissolved oxygen, redox potential, dissolved carbon content, and depth to the water table. Examples of technology-specific performance parameters include drawdown for extraction wells, tracers for enhanced *in situ* bioremediation, and airflow rates for vapor extraction systems. Secondary, tertiary, and technology-specific performance parameters are used in combination with primary metrics to evaluate the efficacy of a remedial system. Circumstantial evidence provided by these types of performance parameters is significant and may be crucial to making the correct ASM decisions. Examples of the use of such data to draw inferences about the performance of a remediation plan are provided in Kampbell et al. (1998), EPA (1998b), Stiber et al. (1999), Wiedemeier et al. (1999), and NRC (2000). These protocols place a special emphasis on data to support the suitability for, and success of, natural attenuation because monitoring is central to implementation of this remedial strategy. However, similar data analysis methods can and should be developed and applied to evaluate the progress of other remediation methods.

Adaptive Monitoring Network Design

The design and implementation of monitoring programs can be made more adaptive to keep data collection activities, as well as the remedial action, as focused and cost-efficient as possible. Drivers for adjusting monitoring programs include changes in site understanding that lead to improved site conceptual models, unexpected monitoring results, alterations in remedial actions, improvements in monitoring technology, and changes in the type of information required by regulations.

In the last several years there have been significant technological advances in decision analysis, field analytics, and data collection technologies for characterization and monitoring work. These present several opportunities for making the characterization and monitoring process more adaptive and more supportive of an ASM approach. They include (in order of maturity and acceptance) (1) enhancing or optimizing existing monitoring networks, (2) incorporating sensors and field analytics in monitoring design, (3) using new technologies for collecting samples such as direct push systems and passive diffusion samplers, and (4) replacing static sampling and analysis plans with dynamic work plans. The following sections discuss each of these potential enhancements to remedial action monitoring programs, providing details on technology maturity and case studies.

Enhancing Existing Monitoring Networks

The first opportunity for adaptive sampling and analysis as remediation proceeds is to allow monitoring locations to be dropped or sampling intervals lengthened in response to monitoring data that show a system performing well. In the same vein, enhancements could involve adding monitoring locations or increasing the sampling frequency for existing locations for a remedial system that shows signs of deteriorating performance. There is often significant financial incentive to use as many existing groundwater wells as possible because of the costs associated with implementing new wells. Monitoring costs come in two forms—the capital cost of installing monitoring systems and the longer-term cost of sampling and maintaining the system. For deep vadose zone systems, installation costs can range into the hundreds of thousands of dollars per well (DOE, 1998). For shallow groundwater systems, these costs may be on the order of tens of thousands of dollars per installation. In any case, capital costs typically dwarf annual sampling costs.

The most widely used method for improving remedial action monitoring network performance is to determine whether monitoring locations need to be changed (i.e., old monitoring locations abandoned or new locations added) or sampling intervals adjusted. A variety of techniques have been suggested for assisting in this process. These techniques include relatively sophisticated fate-and-transport models, geostatistical and time series analyses, and mathematical optimization methodologies as well as relatively simple "rule-of-thumb" techniques.

The optimal design of monitoring networks in surface and subsurface hydrology is a classic problem that has received extensive attention in the scientific literature. Most of the previous work in the groundwater field falls into two categories: networks for site and plume characterization (e.g., Loaiciga et al., 1992) and networks for plume detection at landfills and hazardous waste sites (e.g., Meyer et al., 1994). There has been significantly less work to address questions of remedial action performance evaluation and long-term monitoring—questions that are directly relevant to MDP1 and MDP2 in the ASM process.

Long-term monitoring networks. With the realization that many contaminated sites will not be quickly closed and will thus require long-term monitoring and management, research in monitoring network optimization has shifted toward the objective of reducing long-term sampling costs without sacrificing information gained or protectiveness. The goal of the research published to date is to eliminate data redundancy by identifying a subset of monitoring wells and a reduced sampling schedule that effectively capture a groundwater plume's evolution. Temporal redundancy refers to whether wells are being sampled too frequently, and spatial redundancy refers to whether too many wells are being sampled. An early example that focused on temporal redundancy is the work of Johnson et al. (1996), who were motivated by the observation that in 1993, the laboratory fees alone required for analyzing groundwater samples at the Savannah River Site amounted to nearly $10 million. These researchers developed a simple technique to reduce sampling schedules through analysis of the time series at individual wells. A trial application of their method resulted in an estimated cost savings of $1.8 million at the Savannah River Site.

Several recent studies have combined methods such as fate-and-transport modeling, geostatistics, and optimization to investigate the temporal and spatial redundancy of existing sampling networks (e.g., Cameron and Hunter, 2000; Rizzo et al., 2000). An example of a study that focuses on identifying spatial redundancy in monitoring networks is Reed et al. (2000), which describes a method that combines groundwater fate-and-transport simulation, kriging, and optimization. This method can be used to identify subsets of monitoring wells to sample for producing an estimate of the total mass of the plume mass that is "acceptably close" to that which would result from sampling all of the available monitoring wells. As discussed in Box 3-1, application of this methodology to the Hill Air Force Base indicated that sampling costs could be reduced by nearly 60 percent.

In recognition of the importance of long-term monitoring optimization, several agencies have developed useful formal decision support tools for network design (see the Federal Remediation Technologies Roundtable (FRTR) web site at http://www.frtr.gov/optimization/monitoring/). An example is the MAROS software developed for the Air Force Center for Environmental Excellence, described in Box 3-2 (Aziz et al., 2000). This software package includes (1) parametric and nonparametric statistical analysis of concentration time series, (2) a sampling frequency determination algorithm based upon the "cost effective sampling" method of Ridley and MacQueen (1995), (3) a plume-mapping method, based on Thiessen polygons, that computes the relative importance of each well in estimating the overall average concentration of the plume, and (4) a stepwise optimization that sequentially removes wells that are "redundant" for computing the average plume concentration.

The Navy is clearly interested in optimizing its long-term monitoring systems, as evidenced by the recent development of guidance for the design and evaluation of groundwater monitoring programs (NAVFAC, 2000). This guidance is fairly general in nature, but it does emphasize the importance of annual reviews for monitoring programs, and the potential need for revisiting both remedial strategies and monitoring program design based on the results of those reviews. The guidance suggests various techniques that might be useful in improving monitoring system performance, including basic statistical comparisons, geostatistics, groundwater modeling, and data presentation using geographic information systems (GIS), but it provides little supporting detail.

BOX 3-1
Groundwater Monitoring Optimization at Hill Air Force Base

A BTEX plume previously studied at Hill Air Force Base in Utah was numerically simulated for the purpose of demonstrating the methodology of Reed et al. (2000) for optimizing the choice of monitoring well locations. The areal extent of the two-dimensional, steady-state simulated plume (21,000 m^3) and the locations of 30 potential monitoring wells are shown in Figure 3-15. (Two-dimensional modeling was justified based on the presumed full vertical extent of the plume over the 0.9-m saturated zone.) The total mass of BTEX within a defined subdomain as shown in Figure 3-15 was calculated to be 37.6 kg. Contaminant plume simulation is used to project the migration and mass of BTEX.

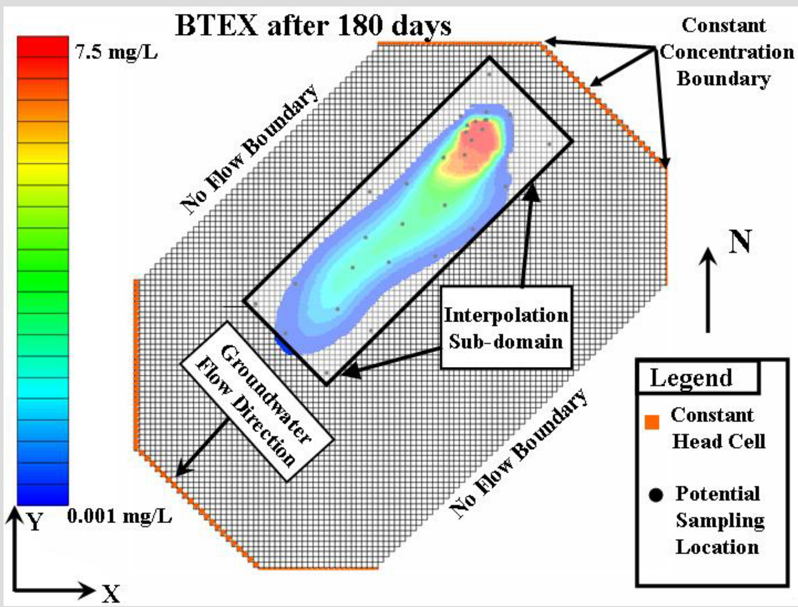


FIGURE 3-15 Simulated BTEX plume and potential monitoring well locations enclosed in the subdomain used for total mass calculations. SOURCE: Reprinted, with permission, from Reed et al. (2000). © (2000) American Geophysical Union.

Continued

BOX 3-1 Continued

Formal mathematical optimization (i.e., a generic algorithm) was used to identify optimal solutions in which a reduced number of sampling points provided accurate mass estimates. Mass estimates were computed using three different approaches for plume interpolation, including kriging, inverse distance weighting, and a hybrid heuristic that uses both of these methods in combination.

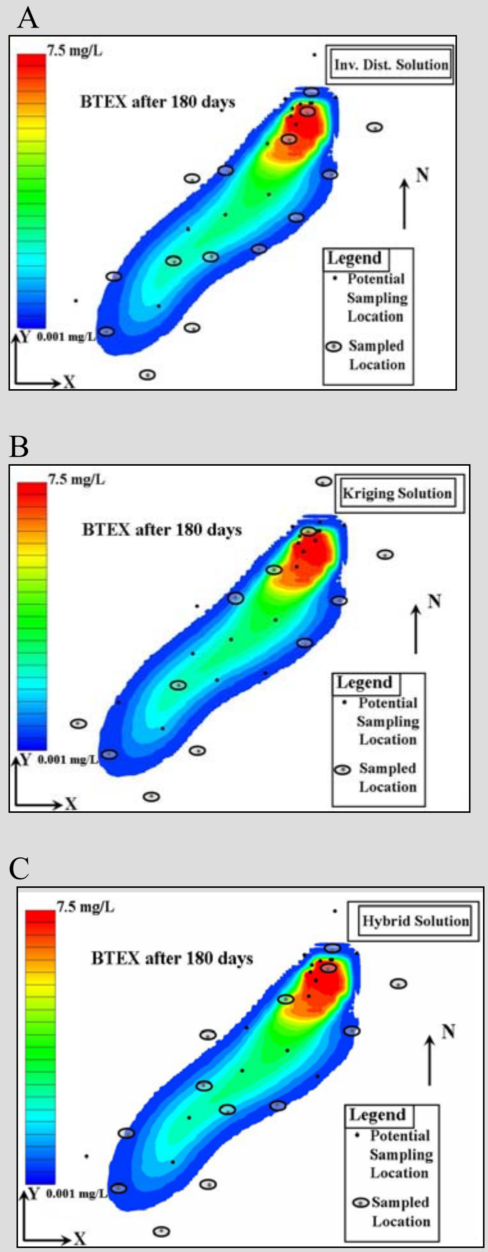
The inverse distance weighting scheme, which is extremely fast computationally, chose an optimal sampling network consisting of 15 wells (Figure 3-16A) and yielded a mass estimate of 46.4 kg. This mass estimate provides nearly the same mass estimate as if 30 wells had been chosen (46.7 kg). Both of these estimates have about 24 percent error compared with the known mass, but the optimal solution would reduce operating and maintenance costs by 50 percent.

The kriging-based optimization solution chose 12 monitoring wells (Figure 3-16B) and estimated the mass of the plume using these wells to be 35 kg, which was also the mass estimated using this method for all 30 wells. This mass estimate was within 7 percent of the known mass, and the total costs were reduced by 60 percent by eliminating 18 monitoring wells. This scheme is more accurate but computationally more expensive and requiring greater technical skill and effort than the inverse distance weighting scheme.

A hybrid solution algorithm combines the best features of the above two approaches. This solution approach chose 13 monitoring wells (Figure 3-16C) and a mass estimate of 35 kg, with a computational time reduced by 67 percent compared to the kriging approach. The mass error is the same as the kriging approach, but the identified solution is not as optimal as that found by the kriging approach because one additional well is required, and therefore the approach represents a tradeoff between computational efficiency and solution cost.

This case study illustrates the beneficial information that can be gleaned from applying mathematical optimization techniques to design a monitoring well system, or to adjust a monitoring well system that is already in place by adding or removing wells. The level of sophistication of a user would be expected to be relatively high owing to the required use of mathematical optimization.

FIGURE 3-16 Optimization of well monitoring networks using three approaches: (A) inverse distance, (B) kriging, and (C) hybrid solution. Note: The highest BTEX concentrations, which are present in the center of the plume, correspond to the top of the concentration scale bars to the left of each figure. SOURCE: Reprinted, with permission, from Reed et al. (2000). © (2000) American Geophysical Union.


A
7.5 mg/L
Inv. Dist. Solution
BTEX after 180 days
N
Legend
Potential Sampling Location
Sampled Location
Y 0.001 mg/L
X
B
7.5 mg/L
Kriging Solution
BTEX after 180 days
N
Legend
Potential Sampling Location
Sampled Location
Y 0.001 mg/L
X
C
7.5 mg/L
Hybrid Solution
BTEX after 180 days
N
Legend
Potential Sampling Location
Sampled Location
Y 0.001 mg/L
X

BOX 3-2
MAROS—The Monitoring and Remediation Optimization System

In recognition of the importance of long-term monitoring optimization, several agencies in the Federal Remediation Technologies Roundtable have developed formal decision support tools for network design (http://www.frtr.gov/optimization/monitoring/). An example is the Monitoring and Remediation Optimization System (MAROS) developed for the Air Force Center for Environmental Excellence (AFCEE) by Groundwater Services, Inc. (Aziz et al., 2000; http://www.afcee.brooks.af.mil/er/rpo.htm).

MAROS is a simple and flexible tool that aims to "optimize" long-term monitoring by adjusting the temporal frequency of sampling and identifying spatially redundant wells. The main information used is the concentration versus time data from the existing monitoring wells for up to five constituents of concern (COCs); these data comprise the so-called "primary lines-of-evidence." Parametric and nonparametric statistical analyses of the time-series trends are used to classify each well and each COC into one of the following categories: decreasing, probably decreasing, stable, increasing, probably increasing, and no trend. MAROS also allows the results of groundwater models and empirical information from various "plume-a-thon" studies to comprise "secondary lines-of-evidence." For example, groundwater models can be calibrated and then used to predict future plume growth. Primary and secondary lines-of-evidence are combined and each monitoring well is classified as to whether it requires "extensive" (i.e., quarterly), "moderate" (i.e., biannually or annually), or "limited" (i.e., annually or biennially) monitoring. For example, if a plume shows a highly confident decreasing trend, then it would be in the "limited" category. A more sophisticated approach to determining sampling frequency is developed in the more advanced MAROS modules. This approach is based upon the so-called "cost effective sampling" method developed by Lawrence Livermore National Laboratory. This approach uses regression to determine the rate of concentration change for individual wells; sampling frequency is based upon this rate, with adjustments for overall long-term trends and compound risk.

A plume-mapping method is used to assess the spatial redundancy of a well. For each well in the network, the concentration is estimated by interpolation with the measured values at nearby wells. (Delaunay triangularization is used as the interpolation method.) Comparison of this estimated value with the true measured value yields a quantitative measure of the importance of the nearby wells. This measure is used in a heuristic optimization step that eliminates well locations that do not contribute significant information about the plume.

Attractive features of the MAROS software include its relatively simple construction and analysis, streamlined data entry and the ability to update data and develop new modules, its use of different levels of reporting ranging from a one-page system optimization summary to individual well trends and statistics, and ability to download the software for free and thus be highly accessible to RPMs. MAROS is best for small to medium sites with fewer than 100 wells. Because MAROS is currently being applied at only a select number of sites, case studies not available at this time.

Boxes 3-3 and 3-4 present case studies of Navy sites where an explicit analysis of monitoring program effectiveness has been performed. Most of the reported case studies (including those in NAVFAC, 2000) focus on the cost savings gained from existing groundwater systems by eliminating redundant monitoring points, reducing sampling frequency, and/or refining analytical lists of contaminants of concern. These cost reductions can be significant, with savings greater than 50 percent over baseline being common. In the context of these case studies, "enhanccing" or "optimizing" existing monitoring systems is synonymous with cost reduction. It is important to note, however, that in ASM the review of monitoring information and monitoring system performance and/or the modification of a remedial strategy could lead to *increased* monitoring requirements and associated costs. The obvious example of this is when a pump-and-treat system is converted to a strategy that relies on monitored natural attenuation.

BOX 3-3
Optimizing Monitoring of the Eastern Plume, NAS Brunswick, Maine

The "Eastern Plume" at NAS Brunswick has resulted from past solvent disposal practices and contains primarily TCE, PCE, 1,1,1-TCA, and limited degradation products. An interim record of decision (ROD) for extraction and treatment was signed in June 1992, and a final ROD for No Further Action for soils and continued pump-and-treat for groundwater was signed in February 1998. The initial groundwater monitoring program included 36 monitoring wells (30 within the plume and six sentinel wells). The monitoring wells were sampled on a triannual basis for VOCs and other compounds. The annual cost for long-term monitoring in 1996 and 1997 was approximately $550,000.

Reviews of the monitoring data showed that the plume was relatively stable. This prompted the Navy to conduct a geostatistical analysis, which revealed some data redundancy as well as data gaps. The Navy met with federal and state regulators and reviewed the records for each sampling location. This resulted in the following key decisions: (1) installation of five new monitoring wells in regions where the data are sparse, (2) reduction of the total number of wells to be sampled from 36 to 22, with 13 in-plume wells and nine sentinel wells, and (3) reduction in the sampling frequency from three to two times per year. Additional cost savings could be realized by modifying the reporting procedures. The annual cost of the monitoring program is anticipated to be approximately $250,000, a savings of over 50 percent.

BOX 3-4
Reducing Sampling Costs in Long-Term Monitoring at NAS Fort Worth (Former Carswell AFB)
Source: HydroGeoLogic, Inc. (2000).

In 1993 Carswell Air Force Based officially closed, and a large portion was transferred to the Navy and renamed the NAS Fort Worth Joint Reserve Base. Activities at the site resulted in the generation of a variety of wastes that have contaminated soil and groundwater. The Air Force, under its Installation Restoration Program, is responsible for cleanup of contamination that occurred prior to October 1, 1993. Most of the effort has focused upon a chlorinated solvent plume, and a pump-and-treat system has been operating to prevent migration of the solvents beyond the eastern boundary of the site. Over 260 monitoring wells have been sampled quarterly at a cost of over $300,000 per year. The plume appears to be relatively stable over time, being effectively contained by the pump-and-treat system.

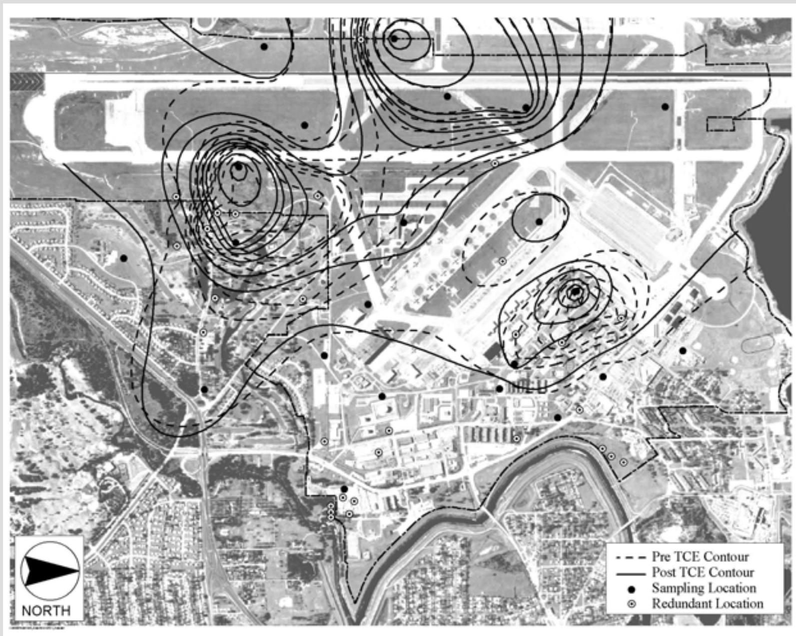
As part of the Year 2000 Groundwater Sampling and Analysis Plan, HydroGeoLogic, Inc., proposed to apply advanced geostatistical techniques to optimize the selection of a subset of wells to be sampled. The application developed by HydroGeoLogic, called the Long-Term Monitoring Optimization (LTMO) Tool Kit, utilizes geostatistical and temporal trending methods to develop sampling plans that eliminate spatial and temporal redundancy. For the NAS Fort Worth site, the objective was to minimize monitoring costs by eliminating sampling locations that do not contribute to characterization of the plume along the eastern boundary of the site. The geostatistical technique known as kriging not only yields a contour map of the contaminant plume, but also gives an estimate of contaminant uncertainty. An "importance factor" for each monitoring well can be calculated based upon its contribution to the overall uncertainty over a region of interest. Monitoring locations with small importance factors are candidates for elimination. In this case, application of the LTMO Tool Kit identified more than 60 percent of the wells as spatially redundant. Because of certain fixed expenses, the overall cost savings realized was somewhat less—the 1999 cost for sampling 193 locations was $447,712, and the 2000 cost for sampling 72 locations was $310,794, resulting in an overall savings of $136,918.

Continued

Incorporation of Sensors and Field Analytics into Monitoring

Within the last ten years there have been significant advances in the quality of field analytical techniques, the number of technologies available, and their regulatory acceptability. Thus, a second opportunity for a more adaptive approach to monitoring within a traditional fixed point monitoring system is to build sensors and/or field analytical methods into the characterization or monitoring process. Field analytics such as test kits or portable instrumentation can be used as a complete substitute for

BOX 3-4 Continued



In this photo of the site, the dashed line contours map the plume prior to optimization and are based upon using all the monitoring locations (the black dots plus the white dots). The solid line contours are post optimization and are based only on the black dots. They demonstrate that the use of fewer wells still maintains a good map of the plume contours along the eastern boundary.

laboratory analyses, or they can augment a laboratory-based program by providing on-the-spot analyses to justify the collection and submittal of samples for more traditional laboratory analyses.

Sensors and field analytics can (1) reduce overall characterization and monitoring costs, (2) provide more complete datasets spatially and over time, and (3) produce more timely results than reliance solely on offsite laboratory analyses. Field analytics and sensors reduce overall characterization and monitoring costs because, in general, the per-sample cost associated with a field analytical analysis is much less than that of the corresponding laboratory analysis. As an example, a field-deployable GC/MS tuned for explosives work was used to support characterization of TNT- and DNT-contaminated soils at Joliet Army Ammunition Plant.

Estimated analytical costs were under $60 per sample for the work, compared to the per-sample cost of approximately $250 for standard offsite laboratory analysis (Johnson et al., 1997).

The use of continuous, or nearly continuous, data collection technologies at fixed monitoring points can provide a much more complete set of data upon which to base performance evaluation decisions. This is partially because the lower costs associated with field analytics for *ex situ* sample analyses can allow a larger number of samples to be collected within the same budget as compared to a traditional monitoring program, providing much more complete coverage spatially and temporally. Off-the-shelf, commercially available, continuous depth-to-water-table measurement systems and data loggers are mature examples of these types of technologies. Technologies for providing continual recording of basic parameters such as temperature and pH have also been available for some time. Advances in sensor miniaturization have recently led to commercially available multiparameter sensors that can simultaneously measure dissolved oxygen, conductivity, and resistivity, along with depth. It is only a matter of time before the range of parameters amenable to *in situ* monitoring expands to include at least some common groundwater contaminants of concern.

The use of either dedicated *in situ* measurement systems or field analytics for rapid in-field sample analyses also provides the opportunity to more quickly identify and respond to potential performance issues with a remediation system. In some cases, such as with natural attenuation of groundwater contamination, system evolution occurs at time scales where rapid identification of changing subsurface conditions is not important. However, for engineered barrier systems and some of the more dramatic intrusive subsurface interventions (e.g., thermal heating, Fenton's Reagent, etc.), quickly identifying unexpected contaminant mobilization or other key potential system failures can be crucial to overall remediation success.

Federal agency research and development programs have heavily invested in the last decade in field analytics and sensor technologies that can be applied to hazardous waste site characterization, remediation, and monitoring activities. For example, DOE's Environmental Management Science Program (http://emsp.em.doe.gov/portfolio/multisearch.asp) currently lists more than 70 research and development projects that address data collection or sample analysis issues. Techniques as diverse as antibody methods, *in situ* microsensors, spectroelectrochemical sensors, spectrometric DNA diagnostics, dielectrics and nuclear magnetic resonance, partitioning tracers, electromagnetic imaging, seismic technologies, acoustic probes, conductive luminescent polymers, cavity ringdown spectroscopy, gamma ray imaging, optical array sensors, noble gas detectors, and BioCOM sensors are mentioned. Likewise, DoD's Strategic Environmental Research and Development Program has funded more than 20 research and development activities focused on characterization and monitoring technologies in its cleanup area. Researchers with the Navy's Space and Naval Warfare Systems Command (SPAWAR) have focused specifically on technologies applicable to the more specialized needs of sediments (see Box 3-5).

In response to these advances, there have been regular modifications to recommended EPA analytical protocols, including SW-846. Within the Resource Conservation and Recovery Act (RCRA) program, EPA's SW-846 contains guidance on acceptable analytical techniques for RCRA-related activities. The latest is Draft IVB (EPA, 2000c), which includes several additions pertinent to Navy contaminants of concern. The EPA Technology Innovation Office (TIO) maintains an encyclopedia of field analytical technologies (http://fate.clu-in.org). The FRTR also maintains a table that provides summary performance information for a wide range of analytical techniques, categorized by contaminant class and media. Table 3-1 provides a summary of field analytical techniques based on the information maintained by the FRTR. In addition, EPA's Environmental Technology Verification program (www.epa.gov/etv)—designed to accelerate the use of innovative technology—has issued reports verifying the validity of over 39 monitoring and characterization technologies. These include, for example, cone penetrometer-deployed sensor technologies, groundwater sampling technologies, PCB field analytical measurement techniques, and portable GC/MS.

Alternative Sample Collection Technologies

Subsurface characterization and monitoring programs have traditionally relied on drilling techniques to obtain soil samples at depth and on permanent, screened and developed monitoring wells for acquiring groundwater samples. Just as there have been advances in field analytical techniques, so too there has been progress made in soil, sediment, and groundwater sample collection technologies. The advantages of these advancements include a reduction in sample collection costs, greater sample production rates, and in some cases more representative samples. In addition, when coupled with field analytical methods, these alternative

BOX 3-5
Rapid Field Characterization of Sediments

Rapid field characterization techniques have been developed to speed assessment and reduce costs. These are field-transportable screening tools that provide measurements of chemical, biological, or physical parameters on a real-time or near real-time basis. Specific advantages include the ability to get rapid results to guide sampling locations, the potential for high data mapping density, and a reduced cost per sample. The approaches do have limitations including the nonspecific nature of some tests, sensitivity to sample matrix effects, and some loss in accuracy over conventional laboratory analyses. A variety of tools have been suggested for the rapid characterization of sediments, as shown in the table below.

Screening-Level Analyses Recommended by the Assessment and Remediation of Contaminated Sediments Program for Freshwater Sediments

Analytical Technique	Parameter(s)
X-ray Fluorescence Spectrometry (XRF)	Metals
UV Fluorescence Spectroscopy (UVF)	Polycyclic Aromatic Hydrocarbons (PAHs)
Immunoassays	PCBs, Pesticides, PAHs
Microtox®	Acute Toxicity

SOURCE: EPA (1994).

The Sediment Management Laboratory of the Space and Naval Warfare Systems Command (SPAWAR), San Diego, CA, has tested the applicability of these characterization technologies for use with sediment, particularly the use of portable XRF to determine metal concentrations (Kirtay et al., 1998; Stallard et al., 1995). The additional spatial resolution afforded by the inexpensive rapid assessment techniques allows a much more thorough characterization of spatial variability at sediment sites and could provide the detailed information necessary for ASM.

sample collection technologies can enable dynamic work plans and adaptive sampling and analysis programs, concepts discussed in the following section.

One example of this innovation is the use of direct push technologies for obtaining subsurface soil, sediment, and groundwater samples. These technologies drive, push, or vibrate small-diameter steel tubes into the ground, up to depths of approximately 100 feet depending on rig type and subsurface lithology. Direct push technologies generally retrieve intact soil cores for *ex situ* sample analysis. With appropriate attachments and modifications, they can also be used to retrieve groundwater and soil vapor samples. Direct push equipment ranges from small, readily transportable units that can be used through floors of buildings, to large dedicated rigs. Box 3-6 describes the adaptation of a direct push technology for use in an estuary environment for rapidly and efficiently retrieving sediment cores.

Direct push technologies can be coupled with field analytics and sensors in a variety of ways to generate pertinent characterization and monitoring data. Properly instrumented direct push rigs can provide information on subsurface lithology through resistivity and stress/strain readings generated by rod advancement. With specialized tips or rod sections, soil, groundwater, and soil vapor samples can be retrieved for *ex situ* analyses. The membrane interface probe (MIP) is an example of a specialized direct push stem design that allows for the near real-time evaluation of subsurface VOC contamination in soils and groundwater when combined with an above-ground detection system such as a photoionization detector or gas chromatograph. Although its detection limits are not sufficient to meet typical groundwater cleanup standards, they are low enough to allow the system to detect the presence of potential subsurface source areas. This type of capability can be extremely useful in refining remedial interventions that target source removal or source degradation.

Specialized direct push tips have been instrumented to support the *in situ* use of x-ray fluorescence (XRF), laser-induced fluorescence (LIF), gamma spectroscopy, and laser-induced breakdown spectroscopy (LIBS) (DOE, 2002). These systems and the data they generate have gained various levels of acceptance by the user and regulatory communities; it is clear from the technical progress to date that they will be widely used in the future. Most work in this area has focused on the generation of pre-remediation characterization information via the DOE and DoD's Site Characterization and Analysis Penetrometer System (SCAPS) programs (EPA, 1995; USAEC, 2000). SCAPS makes use of a cone penetrometer truck to push instrumented tips into the subsurface.

The possibility of rapidly and inexpensively gathering detailed subsurface information in near real time via direct push technologies can change the way remedial action monitoring work is conducted for those settings amenable to direct push technologies. Direct push technologies such as SCAPS can be used to install relatively low-cost *temporary* monitoring points. The combination of direct push with temporary monitoring points allows monitoring to be adjusted cost effectively across space as well as over time in response to data. An obvious example is the temporal tracking of some critical concentration isopleth over time (i.e., the concentration associated with closure guidelines), something that currently is almost impossible to do at most sites using spatially lim-

TABLE 3-1 Summary of Sensor and Field Analytical Techniques

Technique	**Analytes**	***Media***			***Performance***							***Applicable to***			
		Soil/Sediment	**Water**	**Gas/Air**	**Selectivity**	**Susceptibility to Interference**	**Detection Limits**	**Turnaround Time Per Sample**	**Quantitative Data Capability**	**Technology Status**	**Relative Cost Per Analysis**	**Screen Identify**	**Characterize Quantify**	**Cleanup Performance**	**Long-Term Monitoring**
VOC, SVOC, TPH and PCB (*in situ* analysis)															
Solid / Porous Fiber Optic	11	E	A	B	B	A	B	A	B	I	A	A	B	A	B
Laser Induced Fluorescence	5, 11	B	A	NA	B	B	B	A	B	III	A	A	B	A	B
VOC, SVOC, TPH and PCB (*ex situ* analysis)															
Photo-Ionization Detector	1, 3	E	E	A	B	C	B	A	C	III	A	A	C	A	C
Flame-Ionization Detector	1-3	E	E	A	B	C	A	A	C	III	A	A	C	A	C
Explosimeter	1	E	E	A	C	C	B	A	C	III	A	A	C	A	C
Gas chromatography (GC) plus detector	1-6, 11	E	E	A	A	A	A	B	A	III	B	A	A	A	A
Catalytic Surface Oxidation	1,3	E	E	A	B	B	B	A	C	III	A	A	B	A	A
Detector Tubes	1,3	E	E	A	B	B	B	A	C	III	A	A	C	A	C
Mass Spectrometry (MS)	1-6	E	E	A	A	B	B	B	A	II	C	B	A	A	B
GC / MS	1-6	E	E	A	A	A	A	C	A	III	C	A	A	A	B
GC/Ion Trap MS	1-6	E	E	A	A	A	A	B	A	II	C	B	A	A	B
Ion Trap MS	1-6	E	E	A	A	B	A	B	A	II	C	A	A	A	A
Ion Mobility Spectrometer	1-4, 6	E	E	A	A	B	A	A	A	II	B	A	B	A	B
Ultraviolet (UV) Fluorescence	1, 3, 5	B	A	B	C	B	A	B	B	II	B	A	B	A	A
Synchronous Luminescence/ Fluorescence	1-4	E	A	B	B	B	A	B	B	I	B	A	B	A	A

UV-Visible Spectrophotometry	1, 3, 5	E	A	B	C	C	A	A	B	I	B	A	B	A	A
Infrared Spectroscopy	1-4	E	E	A	B	C	B	A	B	II	B	A	B	A	A
Fourier Transform Infrared (FTIR) Spectroscopy	1, 3, 11	E	E	A	A	B	A	A	B	II	B	A	B	A	A
Scattering / Absorption LIDAR	1, 3	E	E	A	C	C	C	A	B	I	B	A	B	A	A
Raman Spectroscopy/ Surface Enhanced Raman Scattering (SERS)	1-5, 11	E	A	E	C	C	A	A	B	II	B	A	B	A	A
Near IR Reflectance/ Transmittance Spectroscopy	1, 3	A	NA	NA	C	C	C	A	B	I	B	A	B	A	A
Immunoassay Colorimetric Kits	1-6, 11	A	A	NA	B	B	B	A	B	II	A	A	B	A	B
Amperometric and Galvanic Cell Sensor	1, 3	E	NA	A	A	B	A	A	B	II	A	A	B	A	A
Semiconductor Sensors	1, 3	E	A	A	B	B	A	A	B	I	A	A	B	A	A
Piezoelectric Sensors	1, 3	E	E	A	A	C	A	A	B	I	A	A	B	A	A
Field Bioassessment	1-6	A	A	A	C	C	NA	C	C	II	C	A	A	C	B
Toxicity Tests	1-6	A	A	A	C	C	NA	B	B	II	A	A	A	C	A
Room-Temperature Phosphorimetry	4, 5, 6, 12 (PCBs)	B	A	B	A	C	A	B	B	I	B	A	B	A	A
Chemical Colorimetric Kits	2, 4, 5, 11	B	A	NA	B	B	B	A	B	II	A	A	B	A	A
Free Product Sensors	11	NA	A	NA	C	A	C	A	C	III	A	A	A	A	A
Ground Penetration Radar	11	B	C	NA	C	C	C	C	B	I	B	B	A	B	B
Thin-Layer Chromatography	2	E	A	NA	B	B	A	B	A	II	C	A	A	A	A
Metals (*ex situ* analysis)															
Atomic Absorption (AA) Spectroscopy	7	E	E	A	A	A	A	C	A	I	C	C	A	C	B

Technique	Analytes	***Media***			***Performance***							***Applicable to***			
		Soil/Sediment	**Water**	**Gas/Air**	**Selectivity**	**Susceptibility to Interference**	**Detection Limits**	**Turnaround Time Per Sample**	**Quantitative Data Capability**	**Technology Status**	**Relative Cost Per Analysis**	**Screen Identify**	**Characterize Quantify**	**Cleanup Performance**	**Long-Term Monitoring**
Metals (*ex situ* analysis)															
Inductively Coupled Plasma-Atomic Emission Spectroscopy (ICP-AES)	7	E	E	A	A	A	A	B	A	I	C	C	A	C	B
X-Ray Fluorescence	7	A	A	E	A	A	B	A	A	III	A	A	B	A	B
Chemical Colorimetric Kits	7, 9	B	A	NA	A	B	B	B	B	II	A	A	B	A	A
Titrimetry Kits	7, 9	B	A	NA	A	B	B	B	B	III	A	A	B	A	A
Immunoassay Colorimetric Kits	7, 12 (Hg)	A	A	NA	B	B	A	A	B	II	A	A	B	A	A
Anodic Stripping Voltammetry	7	E	A	NA	A	B	A	A	A	II	B	B	A	A	B
Fluorescence Spectrophotometry	7, 12 (Hg)	E	E	A	A	B	A	A	A	II	B	A	A	A	A
Amperometric and Galvanic Cell Sensor	7	E	A	NA	A	B	B	A	B	II	A	A	B	A	A
Field Bioassessment	7, 9	A	A	A	C	C	NA	C	C	II	C	A	A	C	B
Toxicity Tests	7, 9	A	A	A	C	C	NA	B	B	II	A	A	A	C	A
Ion Chromatography	7	E	A	NA	B	B	A	A	A	I	B	A	A	A	A
Explosives (*ex situ* analysis)															
Gas chromatography (GC) plus detector	10	E	E	B	A	A	B	B	A	II	C	B	B	B	B
Mass Spectrometry	10	E	E	B	B	C	B	A	B	II	C	B	B	B	B

GC / MS	10	E	E	A	A	A	B	B	A	II	C	B	B	B	B
Ion Mobility Spectrometer	10	E	E	A	A	C	A	B	A	I	C	A	C	C	C
Field Bioassessment	10	A	A	A	C	C	NA	C	C	II	C	A	A	C	B
Toxicity Tests	10	A	A	A	C	C	NA	A	B	II	A	A	A	A	A
Chemical Colorimetric Kits	10	E	A	NA	B	B	B	A	B	III	A	A	B	B	A
Immunoassay Colorimetric Kits	10	E	A	NA	B	B	B	A	B	III	A	A	B	B	A

Legend:

Media and/or Applicable To	A	Better	B	Adequate	C	Serviceable
	NA	Not applicable	E	Requires selection of extraction procedure		
Selectivity	A	Measures the specific contaminant directly	B	Measures the contaminant indirectly	C	Measures a part of the compound
Susceptibility to Interference	A	Low	B	Medium	C	High
Detection Limits	A	Low: 100-1000 ppb (soil); 1-50 ppb (water)	B	Midrange: 10-100 ppm (soil); 0.5-10 ppm (water)	C	High: 500+ ppm (soil); 100+ ppm (water)
	NA	Not applicable				
Turnaround Time per Sample	A	Minutes	B	Hours	C	More than a day
Quantitative Data Capability	A	Produces quantitative data	B	Data is quantitative with additional effort	C	Does not produce quantitative data
Technology Status	III	Commercially available and routinely used field technology				
	II	Commercially available technology with moderate field experience				
	I	Commercially available technology with limited field experience				
Relative Cost per Analysis	A	Least expensive	B	Mid-range expensive	C	Most expensive

Analytes

1- Non-halogenated volatile organics
2- Non-halogenated semivolatile organics
3- Halogenated volatile organics
4- Halogenated semivolatile organics
5- Polynuclear aromatic hydrocarbons (PAHs)
6- Pesticides / herbicides
7- Metals
8- Radionuclides
9- Other inorganics (asbestos, cyanide, fluorine)
10- Explosives
11- Total petroleum hydrocarbons
12- Specific analyte (named in matrix)

SOURCE: Adapted from FRTR (http://www.frtr.gov/site/analysismatrix.html).

BOX 3-6
Hoverprobe Sediment Coring and Water-Quality Profiling

Many Navy and other DoD facilities are located adjacent to surface-water bodies where plumes may discharge to locations such as wetlands and estuaries that are relatively difficult to access and that contain habitat sensitive to the disturbances caused by traditional drill rigs. These difficulties limit the technologies available to obtain necessary hydrogeologic and water-quality information for site characterization and optimization of groundwater monitoring networks. In response to these needs, a unique drilling and water-quality profiling system, mounted on a hovercraft and called the "Hoverprobe 2000," was developed by the U.S. Geological Survey in cooperation with Hovertechnics, Inc., of Benton Harbor, Michigan, and MPI Drilling, Inc., of Picton, Ontario (Phelan et al., 2001). A hovercraft is a versatile vehicle that can be propelled over the surface of land, water, mud, snow, or ice by a cushion of air produced by downwardly directed fans. It can also be landed on the surface of these difficult terrains and proceed to or from a submerged site even if insufficient water is present to float it. A segmented skirt constructed of rubber-coated fabric surrounds the base of the craft and traps most of the pressurized air under the craft. At rest, the Hoverprobe exerts a pressure that is about 10 percent of the pressure exerted on the ground by a standing person, allowing drilling and sampling in wetlands and tidal flats with minimal surface disturbance (Phelan et al., 2001). The vibracore drill on the Hoverprobe uses hydraulically driven cams to generate high-frequency vibrations to drive casing into the subsurface without use of drilling fluids and with almost no cuttings resulting at the surface. The Hoverprobe can be used for the collection of sediment cores, for drive-point water-quality profiling similar to direct push sampling technologies, or for installation of monitoring wells. Continuous sediment cores can be obtained to a depth of about 100 ft from saturated unconsolidated sediments. Drilling and sampling can occur while the craft is on mud or on solid ground or is floating on water, and can continue as water levels or tides shift.

The first use of the Hoverprobe in a groundwater contamination investigation was as part of an evaluation of natural attenuation of chlorinated solvents discharging to freshwater tidal wetlands and creeks at Aberdeen Proving Ground, Maryland. Although monitored natural attenuation has been shown to be a feasible groundwater remediation method for chlorinated solvents discharging to the tidal wetland and creek (Lorah et al., 1997, 1999a,b), the acceptance of a remediation strategy was delayed by the lack of definition of the southern extent of the plumes discharging to the tidal creeks and of the hydrogeology of the creek channel. Regulators were concerned that subsurface migration of contaminants could occur downstream beneath the creek channel, transporting contaminants to an estuary of the Chesapeake Bay without discharge through wetland sediments where biodegradation of the chlorinated solvents occurs. The Hoverprobe allowed investigation at 13 sites along the stream channel that were previously inaccessible because of mud and shallow water (Phelan et al., 2001). Continuous sediment coring and water-quality profiling for chlorinated volatile organic compounds and redox-sensitive constituents were conducted without installation of wells, providing data to define plume boundaries and to refine the hydrogeologic parameters in a groundwater flow model used to assist in evaluating remedial alternatives.

The Hoverprobe and a support hovercraft during drilling and water-quality profiling along the West Branch, Canal Creek, Aberdeen Proving Ground, MD. The support hovercraft was used for transport of samples to nearby laboratory facility for immediate analysis and in case emergency exit was needed.

ited monitoring well information. Although these types of technologies may never be appropriate for deep vadose zone sites or sites with fractured rock flow systems, they would be appropriate for the majority of coastal Navy facilities with relatively near-surface saturated zones and contamination events.

In the case of traditional monitoring wells, techniques for obtaining less expensive and more representative groundwater samples have also been developed. These include low purge technologies and passive diffusion samplers. Passive diffusion samplers can eliminate altogether the need for purging monitoring wells before sampling. Diffusion samplers are a class of samplers, developed by Don Vroblesky at the U. S. Geological Survey, that are based on the laboratory and field confirmation that VOCs can diffuse through low-density polyethylene films and reach equilibrium concentrations that correlate well with actual subsurface contaminant concentrations (USGS, 2001). Types of diffusion samplers include water-to-water samplers and vapor-to-vapor samplers. Both types are applicable to the sampling of groundwater (via wells), the groundwater/surface water interface, pore water in sediments, surface water, and water from treatment systems. Vapor-to-vapor samplers are also effective for measuring *in situ* soil gas and vapor phase concentrations in confined spaces.

The effectiveness of diffusion samplers is dependent upon the samplers being in direct contact with volatile organic compounds. Diffusion samplers should not be deployed in monitoring wells where sand packs are less permeable than the surrounding formation. In addition, diffusion samplers are not recommended for the quantitative measurement of methyl-tertiary butyl ether (MTBE) or acetone.

Multiple diffusion samplers deployed in a vertical array can provide an effective method of vertical contaminant profiling in monitoring wells. Optimal conditions would consist of the diffusion sampler or groundwater monitoring well screen being in direct contact with the surrounding formation, but correctly designed monitoring well sand packs are also appropriate. The presence of vertical gradients across the sampling interval will compromise the resolution of vertical contaminant profiling.

The most promising application for diffusion samplers appears to be for long-term groundwater monitoring in wells, with the potential to reduce long-term monitoring costs by 20 percent to 50 percent. Detailed information regarding the appropriateness, construction, deployment, handling, and analysis of diffusion samplers can be found in USGS (2001).

Dynamic Work Plans

The last opportunity for developing a more flexible and adaptive approach to subsurface performance monitoring is to base a characterization or monitoring program on dynamic work plans. Dynamic work plans differ from more traditional sampling and analysis plans in that they identify the decision logic that will be used for determining the appropriate analytical techniques and sample numbers, locations, and frequency *as work proceeds*, rather than pre-specifying those data collection characteristics. As alluded to above, dynamic work plans rely at least in part on direct push technologies and field analytic techniques. With these technologies, data collection can be adapted in response to the changing information needs of a remedial action, and the remedial action itself can be adjusted or adapted based on feedback from the data collection.

The concept of developing hazardous waste site characterization programs based on dynamic work plans has been implemented under a variety of names, including expedited site characterization (DOE, 1998) and adaptive sampling and analysis programs (DOE, 2001). The EPA TIO has been advocating the Triad approach (EPA, 2001) to environmental data collection, which adds systematic planning to the dynamic work plan/field analytic mix. The EPA Superfund program is currently preparing draft guidance on the development of dynamic work plans. Case studies that document characterization cost reductions associated with these types of approaches usually report savings on the order of 50 percent or more. These savings are derived from reductions in per-unit analytical costs and in the overall number of samples collected.

Although the emphasis has historically been on site characterization, dynamic work plan concepts and associated technologies (field analytics, sensors, direct push, etc.) are equally applicable to the remediation phase of site restoration. In fact, the potential impacts on overall costs and remediation performance are greater during remediation than they are during characterization because savings can be realized both from reductions in data collection costs and from improved remedial action performance. In this context, dynamic work plans are a natural component of ASM.

Dynamic work plans are particularly applicable to contaminated soil excavations or contaminated sediment dredging operations. Box 3-7 describes the adaptive nature of a removal project for soils contaminated with radionuclides. A similar example, but in the context of pesticide-contaminated soils, was reported in USACE (2000). In this example,

BOX 3-7
Precise Excavation at the Ashland 2 Site

The U.S. Army Corps of Engineers (USACE) is conducting cleanup of radiologically contaminated properties as part of the Formerly Utilized Sites Remedial Action Program (FUSRAP). The largest cost element for most of the FUSRAP sites is the excavation and disposal of contaminated soil. Conventional approaches to the design of soil excavation/disposal programs delineate excavation boundaries based on existing characterization data. Excavation then proceeds using these design drawings as the basis for determining which soil must be excavated and which can remain. There is considerable evidence that in fact most pre-remediation characterization datasets are inadequate for precisely delineating contamination footprints. The result can be overexcavation of clean soil at considerable unnecessary expense.

A precise excavation approach was implemented at the Ashland 2 FUSRAP site. Data collection was embedded into the excavation program, with data collection consisting of real-time *in situ* sensors, global positioning system units, and an onsite laboratory. Excavation work proceeded in lifts that ranged from 0.5 to 2 feet in depth, with dig-face screening occurring before excavation continued. A pre-excavation estimate of contaminated soil volumes based on RI/FS data placed the total at 14,000 cubic yards. By the time the work was completed, approximately 45,000 cubic yards of soil were identified as being contaminated at levels that were above the cleanup criteria and were excavated for offsite disposal.

A post-excavation analysis specifically of the initial surficial lift showed that if excavation of surficial soil had been based solely on pre-existing data, it would have removed 4,000 cubic yards of minimally contaminated soil (i.e., where soil contaminant concentrations were below the cleanup criteria), and it would have missed 8,000 cubic yards of soil that had contamination is excess of the cleanup

immunoassay kits were used to better define excavation footprints and verify dig-face cleanup guideline compliance at the Wenatchee site. In its cost and performance report, the USACE indicated that overall remediation costs were half of what would have been incurred if excavation had proceeded on the basis of existing historical datasets alone.

There is also a place for dynamic work plans within groundwater remedial action monitoring. A simple example is a plan that samples a traditional network of monitoring wells. In this instance a dynamic work plan might rely on passive diffusion samplers for generating samples and on field analytics for screening those samples. Based on the results, a decision might be made to replicate analyses using an offsite laboratory, to expand sampling to adjacent wells that would not have otherwise been

criteria. Preliminary cost estimation work indicated that the additional cost of the excavation support data collection was approximately $168,000 over six months of excavation. Over $1.5 million in cost savings were achieved by avoiding unnecessary offsite disposal costs for just the initial surficial lift (Durham et al., 1999).

sampled in that round, or to increase sampling frequency in the short term. In the situation where a technology such as direct push was available for quickly acquiring groundwater samples from new locations, or for installing temporary monitoring points, the decision might be to expand the network in the short term to address unexpected trends or results in datasets.

Alternatively, a monitoring system might include real-time data acquisition from dedicated *in situ* sensors. A dynamic work plan would identify the types of result scenarios that would require a response, either by requiring additional data collection or by revisiting the remedial system. An example would be real-time monitoring of a leachate collection system for parameters that might indicate a containment cell failure. A second example would be continuous depth-to-water-table sensors positioned around a barrier wall whose relative potentiometric results might indicate loss of groundwater capture. These latter examples do not represent current practices for monitoring system design, but they do suggest ways that dynamic work plans and adaptive sampling techniques could be used to facilitate an ASM approach to remedial action performance evaluation.

MAJOR CONCLUSIONS AND RECOMMENDATIONS

This chapter was meant to provide general guidance on how to assess remedial performance monitoring with graphical tools and on some of the new monitoring tools available to do so. A major challenge in implementing adaptive site management will be to design the information-gathering efforts to support the management decision points fleshed out in Chapter 2. Thus, monitoring plans should be developed from clearly articulated objectives (such risk reduction, reduction in some indicator of risk, or mass removal), they should support the evaluation of remedial operations performance (MDP2), and they should validate or refine site conceptual models. More specific recommendations that link monitoring to the ASM process are provided below.

Plots of mass removal or concentration versus time or cost (or other metrics depending on the remedy) are objective and transparent tools for illustrating remedial effectiveness that should trigger when to either modify or optimize the existing remedy or to change the remedy. Such graphs should be used after remedy selection to address management decision periods 2 and 3 of ASM. Graphical representations should serve both to enhance stakeholder understanding of the options and to make better decisions about implementing or modifying remedies. At individual sites under investigation, the Navy, in consultation with all stakeholders, should select a unit cost for the continued operation of the remedial action, above which the existing remedy is no longer considered a tenable option.

The Navy should collect and analyze data to develop and validate predictive models of remedy performance. The remedy selection process could be made more quantitative and transparent with the provision of design guidance, charts, and models that summarize technology applications and predict their performance in different environmental settings.

Uncertainties in hydrogeologic data, contaminant concentrations, and rates of remediation should be explicitly recognized in the development and application of performance plots. There are many sources of uncertainty in estimating the mass or risk reduction achieved by any remediation scheme. When sufficient site data are available, statistical methods can be used to estimate error or confidence bands on the performance plots. Site monitoring plans should be developed to ensure that the collected data serve to reduce uncertainty.

A concerted effort should be made to increase monitoring program effectiveness (and to reduce costs) by optimizing the selection of monitoring points, incorporating field analytics and innovative data collection technologies such as direct push, and adopting dynamic work plans and adaptive sampling and analysis techniques. Real-time *in situ* monitoring technologies should also be considered as they mature. These techniques enhance the collection of information upon which ASM decision making is based. DoD should continue to support and foster research in chemical, physical, and biological techniques that would provide more rapid and adaptive approaches for monitoring remedy effectiveness.

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